ELECTRIC POWER RESEARCH INSTITUTE

White Paper: Model Evaluation of the Relative Performance of Alternative Liners

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Abbreviations

CCR	Coal Combustion Residual
CSM	Conceptual Site Model
DC	District of Columbia
EPACMTP	United States Environmental Protection Agency Composite Model for Leachate Migration and Transformation Products
ft AMSL	Feet Above Mean Sea Level
ft bgs	Feet Below Ground Surface
GCL	Geosynthetic Clay Layer
GM	Geomembrane
HDPE	High-Density Polyethylene
HHE	Human Health and the Environment
JRWPP	JR Whiting Power Plant
MCL	Maximum Contaminant Level
NPDES	National Pollutant Discharge Elimination System
POC	Point of Compliance
POE	Point of Exposure
RCRA	Resource Conservation and Recovery Act
RSL	Regional Screening Level
SI	Surface Impoundment
TDS	Total Dissolved Solids
US EPA	United States Environmental Protection Agency
USGS	United States Geological Survey

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1 Background

1.1 Introduction

In 2015, the United States Environmental Protection Agency (US EPA) published a final rule to regulate the disposal of coal combustion residuals (CCRs) in surface impoundments (SIs) and landfills (US EPA, 2015). This rule (hereafter called "the CCR Rule") established criteria for liner systems to determine whether a CCR unit was lined or unlined. A lined/unlined classification is important for SIs because unlined SIs are subject to unique closure requirements. Under the CCR Rule as originally promulgated, a lined CCR unit was required to have one of the following types of liner systems:

- "A liner consisting of a minimum of two feet of compacted soil with a hydraulic conductivity of no more than 1×10^{-7} cm/sec" (40 CFR 257.71[a][1][i]);
- A composite liner system with "the upper component consisting of, at a minimum, a 30-mil geomembrane liner (GM), and the lower component consisting of at least a two-foot layer of compacted soil with a hydraulic conductivity of no more than 1×10^{-7} centimeters per second (cm/sec). GM components consisting of high density polyethylene (HDPE) must be at least 60-mil thick" (40 CFR 257.70[b]); or
- An alternative composite liner system with "the upper component consisting of, at a minimum, a 30-mil GM, and a lower component, that is not a geomembrane, with a liquid flow rate no greater than the liquid flow rate of two feet of compacted soil with a hydraulic conductivity of no more than 1×10^{-7} cm/sec. GM components consisting of high density polyethylene (HDPE) must be at least 60-mil thick" (40 CFR 257.70[c]).

On August 21, 2018, the District of Columbia (DC) Circuit Court vacated portions of the CCR Rule relating to liner requirements. The Court found that there was a lack of support for US EPA's position that potential leaks from clay and alternative liner systems could be detected and promptly addressed, and would thus meet the Resource Conservation and Recovery Act (RCRA) standard that they can be protective of human health and the environment (HHE). Furthermore, the Court determined that only composite liners met the RCRA HHE protection standard (US Court of Appeals, 2018). The Court's decision was based on information contained in US EPA's 2014 "Human and Ecological Risk Assessment of Coal Combustion Residuals" report (herein referred to as "the CCR Risk Assessment"; US EPA, 2014), which indicated that clay-lined SIs were more likely to leak than composite-lined SIs. Specifically, citing the regulatory impact analysis (US EPA, 2010a) performed by US EPA, the Court said that "clay-lined surface impoundments have a 9.1 per cent chance of causing groundwater contamination at drinking water wells at a one-mile distance from the impoundment perimeter" (US Court of Appeals, 2018). Implicitly, this statement indicates that 90.9% of the clay liner scenarios modeled in the CCR Risk Assessment had maximum concentrations lower than the criteria for contamination at a drinking water well at a 1-mile distance, suggesting that a clay liner, as modeled by US EPA, may be protective in some circumstances. However, an evaluation of alternative liner performance that would better facilitate comparing different scenarios would be to assess impacts to groundwater at a fixed point of compliance (POC, i.e., the downgradient boundary of the waste facility) rather than at a variable point of exposure (POE, i.e., distance of 1 mile) as US EPA did for the 2014 CCR Risk Assessment (US EPA, 2014). US EPA's approach in the CCR Risk Assessment may result in the assessment of groundwater impacts at POEs that are both located a

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considerable distance from the waste facility boundary and not located in the most-concentrated portion of the plume.

1.2 Objective

The objective of this research is to evaluate the performance of alternative liners relative to a base case composite liner as specified under the CCR Rule. If alternative liners are capable of performing similarly to the base case liner, then that provides evidence that that those alternative liners *can be* protective of HHE, in which case a performance standard approach to regulating facilities with alternative liners can be protective of HHE. Conversely, if alternative liners do not perform similarly to the base case, then a performance standard is less likely to be protective of HHE. A three-phased approach was used to address this objective:

- A review of the conceptual differences between different liner alternatives (described in a separate white paper; Benson, 2019).
- A model evaluation similar in approach to the CCR Risk Assessment, but with some modifications to facilitate relative evaluation of alternative liners at the POC. Sections 2, 3, and 4 of this white paper describe the modeling and comparative results for SIs.
- A case example in which a detailed review was performed on an SI with an alternative liner for which no statistically significant increases in concentration have been determined to-date (Section 5 of this white paper).

1.3 Approach

The research team developed analytical, empirical, and probabilistic models to simulate the infiltration of leachate through a base case composite liner as specified in the CCR Rule and several alternative liners, as well as the resulting downgradient migration in groundwater. The approach was generally consistent with the information reported in US EPA's CCR Risk Assessment (US EPA, 2014), including using the US EPA Composite Model for Leachate Migration and Transformation Products (EPACMTP) modeling software. The model was used to develop probabilistic distributions of concentrations for two selected inorganic constituents in downgradient groundwater for each modeled liner alternative. These distributions were used to evaluate the different liner alternatives' relative risks to HHE.

Section 2 of this white paper presents the conceptual site models (CSMs) used in the modeling to represent CCR SIs, the alternative liners evaluated, and the hydrogeologic environments simulated in the modeling. Section 3 describes the modeling approach, which included analytical and empirical calculations of infiltration through various alternative liners as inputs to the EPACMTP model. Section 4 presents and discusses the modeling results; the model results are relative and do not represent specific or expected concentrations at any SI site. Furthermore, the model results used for this evaluation are conservative, as explained in Section 3. Sensitivity analyses testing some of the assumptions used in the modeling are also presented in Section 4. Section 5 presents a case study of an SI in southeastern Michigan that evaluates the adequacy of the site-specific geological characterization and the groundwater monitoring well network to determine whether the SI's alternative, natural clay liner is performing effectively. Section 6 summarizes the conclusions presented in this white paper.

The parameter values and modeling approach used in this research were chosen to be consistent with US EPA's CCR Risk Assessment (US EPA, 2014), when feasible. The use of particular parameter values and/or models should not be viewed as a technical acceptance or endorsement by EPRI or Gradient.

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2 Conceptual Site Models

This section presents the CSMs that were developed in order to construct the modeling scenarios. The CSMs provide details on the SIs, the liner scenarios, and the hydrogeological conditions underlying the SIs. These CSMs were designed to be as consistent as possible with the CSMs designed by US EPA in the Agency's CCR Risk Assessment (US EPA, 2014).

2.1 Surface Impoundments

SIs are waste management units in which the release of leachate is driven by ponded water in the impoundment (US EPA, 2003a, p. xxviii). An SI may be located entirely above the ground surface, surrounded by natural embankments, valley walls, and/or engineered berms, or may be wholly or partially incised into the ground. When managed in SIs, CCRs are typically sluiced to the SI, where the solid components settle and the liquid components evaporate, infiltrate, decant into a secondary impoundment or National Pollutant Discharge Elimination System (NPDES)-permitted outfall, or are stored in a pond. Infiltration into the subsurface can occur vertically through the bottom of the impoundment. Natural and/or engineered liners may be used at SIs to restrict infiltration. Figures 2.1 through 2.4 show CSMs of infiltration from SIs under the different liner alternatives considered in this modeling evaluation, which are described in Section 2.2.

An important criterion that may influence the model results is the depth of groundwater at the SI site prior to and after SI operation. SIs constructed completely above the groundwater table are referred to as having "non-intersecting groundwater conditions," and SIs constructed into the ambient groundwater table are referred to as having "intersecting groundwater conditions." Because this research is presenting a relative comparison between different liner types, all the SIs modeled were assumed to have non-intersecting groundwater conditions, i.e., the SIs have an incised depth less than or equal to the pre- and post-development depth to groundwater, consistent with US EPA (2014, p. 4-9). Furthermore, due to modeling restrictions in the US EPA software package, SI characteristics were selected such that the size of the groundwater mound produced by the SI does not intersect the bottom of the SI (US EPA, 2003a, p. 2-39 to 2-41).

2.2 Liner Scenarios

SIs have historically been constructed with a number of different liner types designed to restrict infiltration. The types of SI liners include both engineered and natural systems. Engineered systems include single-layer clay liners and multi-layer composite liners. Natural liner systems are used at SIs that are constructed in a geologic environment, often by design, that acts as a barrier to infiltration. A description of each liner that was evaluated in this research is provided below and in Table 2.1.

- **Unlined:** The unlined scenario represents an SI built directly on native soil that is not necessarily a barrier to infiltration. Similar to the approach used by US EPA (2010b), the analysis in this research assumes that CCR material from the unlined SI migrates vertically downward to create a clogged soil layer. The clogged soil is assumed to have a hydraulic conductivity equal to 10% of the underlying soil conductivity and a thickness of 0.5 m (US EPA, 2003a, p. 2-28). A CSM of the

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unlined impoundment used in this modeling analysis is shown in Figure 2.1. Note that the unlined scenario is not considered an alternative liner for this research.

- **Engineered Clay Liner:** This scenario represents an SI that is constructed with a single-lined, low-hydraulic conductivity clay layer at the bottom of the impoundment. Modeling evaluations for this research assumed an engineered clay liner that is 3 ft thick with a hydraulic conductivity of 10^{-7} cm/s. A CSM of the engineered clay liner used in this modeling analysis is shown in Figure 2.2.
- **Natural Clay Liner:** This scenario represents SIs that are constructed in geologic environments with naturally low hydraulic conductivities that provide a natural barrier to infiltration. Modeling evaluations for this research assessed natural clays that are 25 ft, 75 ft, and 150 ft thick, with hydraulic conductivities of 10^{-7} and 10^{-8} cm/s. In addition, a natural clay liner scenario was also modeled to reflect conditions at the case study site that is discussed in Section 5; this scenario included a 35-ft-thick natural clay liner with variable hydraulic conductivity, ranging from 5.5×10^{-9} to 2.2×10^{-8} cm/s. A CSM indicating how natural clay liners were simulated in this modeling analysis is shown in Figure 2.3.
- **Composite Liner:** This scenario represents SIs constructed with multi-layer composite liners. Typically, composite liners consist of two layers – an upper geomembrane (GM) layer, such as high-density polyethylene (HDPE), and an underlying secondary layer. Because GMs are effectively impermeable, leakage primarily occurs through membrane defects or pinholes, which may occur during manufacturing, installation, or due to weathering. Specific composite liner modeling evaluations presented in this research include:
    - The base case composite liner scenario consists of a 60-mil HDPE GM with defects occurring at an average frequency as defined by a nationwide distribution used by US EPA (2003b, p. 2-23) and an underlying secondary 2-ft-thick, compacted soil layer with a hydraulic conductivity of 10^{-7} cm/s. Simulations were performed for both good and poor contact between the GM and the compacted soil layer. "Good contact" is when the GM has few wrinkles and is installed overlying a smooth secondary soil layer; "poor contact" is when the GM has many wrinkles and is installed overlying a rough secondary soil layer. These liners meet the federal liner design criteria specified in 40 CFR 257.70(b).
    - Because the presence of defects in a GM may be related to its thickness, the performance of a thinner (e.g., 40-mil), non-federally compliant composite liner was simulated by increasing the defect frequency. This is a conservative approach; URS (Andrews and Loellen, 2008) performed a literature review and found no evidence of a correlation between GM thickness and defect frequency:

      "No report mentions, nor is there any reason to believe, that the thickness of the GM would affect the number of defects occurring in a GM in a cap or liner system immediately following installation. This necessarily assumes a proper design, an adequate cushion layer, an appropriate subgrade specification, and a properly designed and implemented maintenance plan. Except for extraordinarily thick GM[s], the causes of the defects (i.e., mechanical damage, faulty welds and seams, stones, and construction worker-derived) are significantly severe that damage would result regardless of the GM type or thickness." (Andrews and Loellen, 2008)

      Furthermore, a companion white paper (Benson, 2019) to this document qualitatively observes that a thinner geomembrane will be more flexible, potentially resulting in better contact with the substrate, than a thicker geomembrane of the same material, which can result in reduced leakage that – conceptually – offsets increased potential for perforations.

- A 60-mil HDPE GM with an underlying geosynthetic clay layer (GCL) was also evaluated. The GCL was assumed to be 0.01 m thick and have a hydraulic conductivity of 10^{-9} cm/s.

A CSM indicating how composite liners were simulated in this modeling analysis is shown in Figure 2.4.

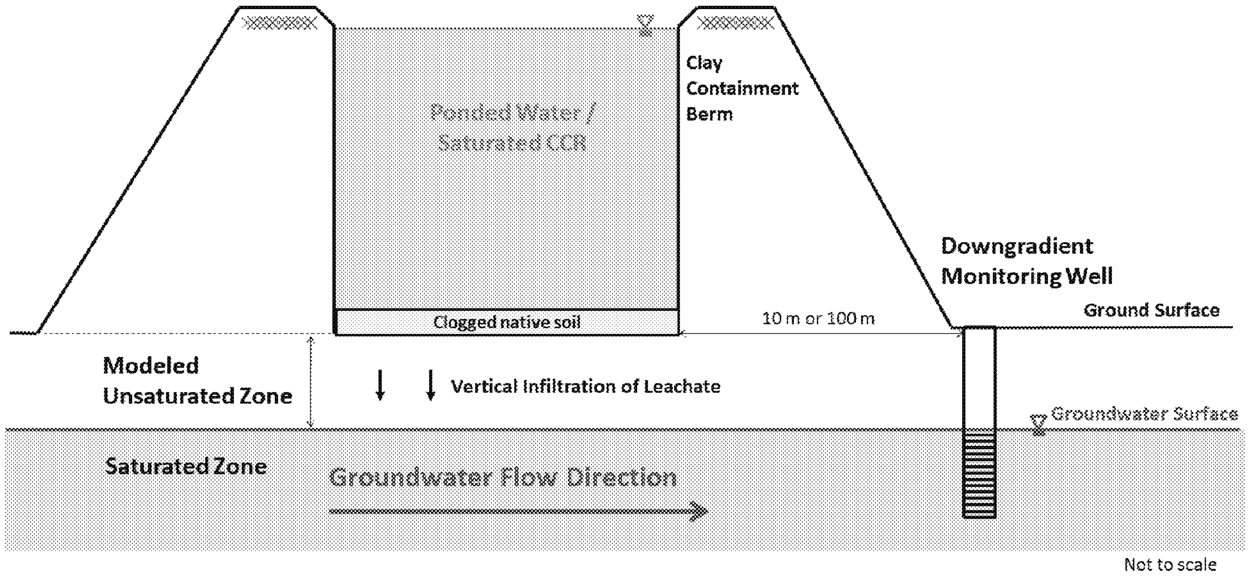

Figure 2.1 Unlined Surface Impoundment Conceptual Site Model

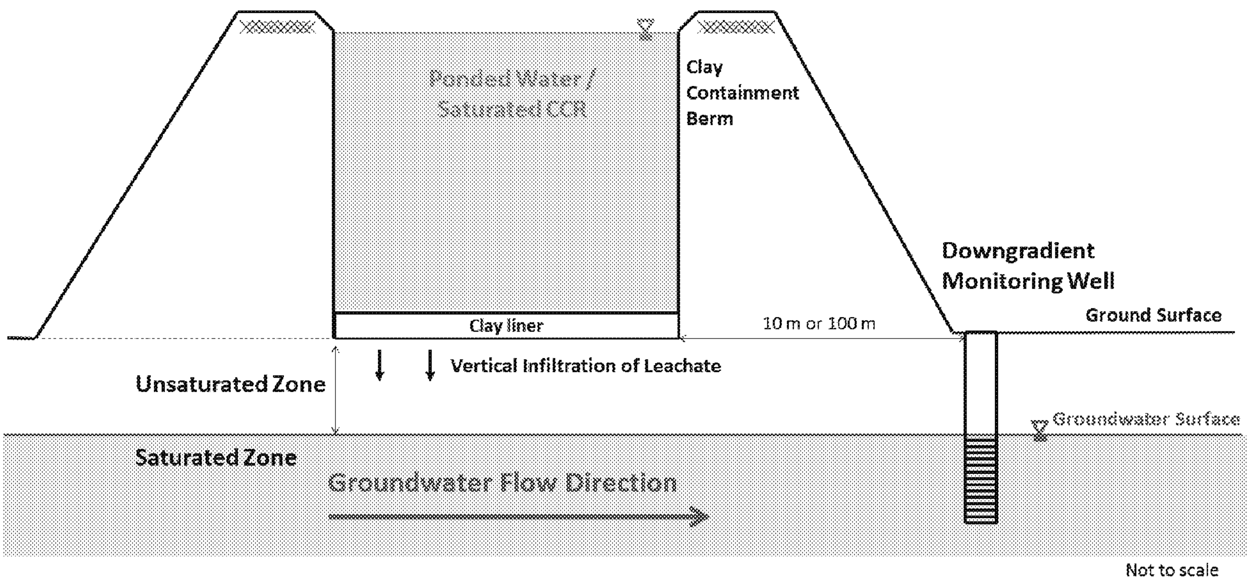

Figure 2.2 Engineered Clay-Lined Surface Impoundment Conceptual Site Model

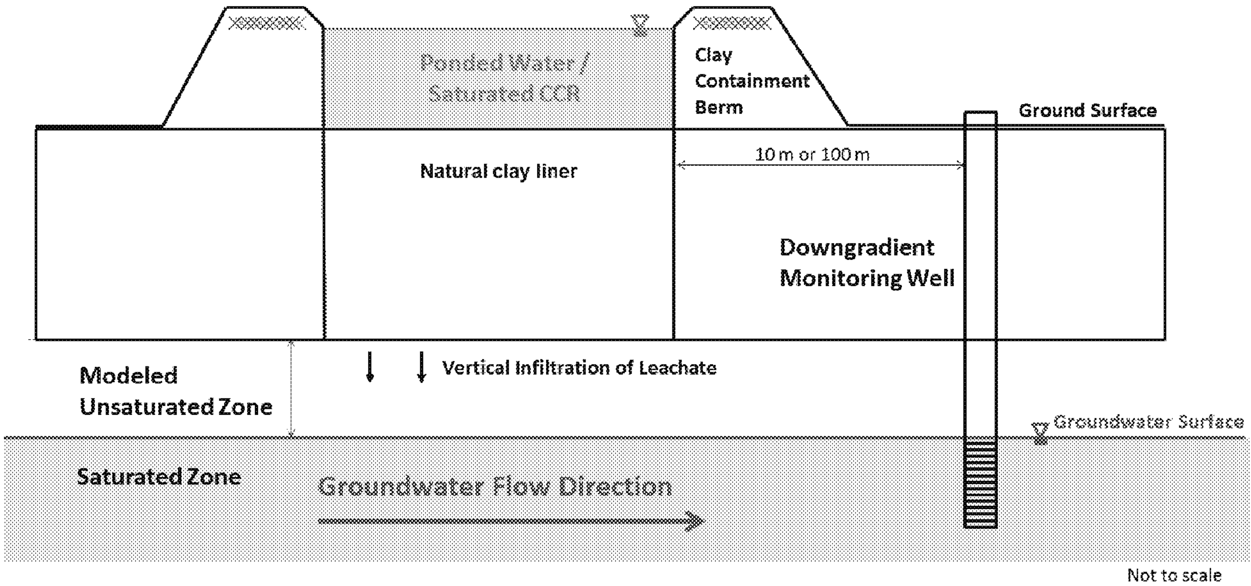

Figure 2.3 Natural Clay-Lined Surface Impoundment Conceptual Site Model

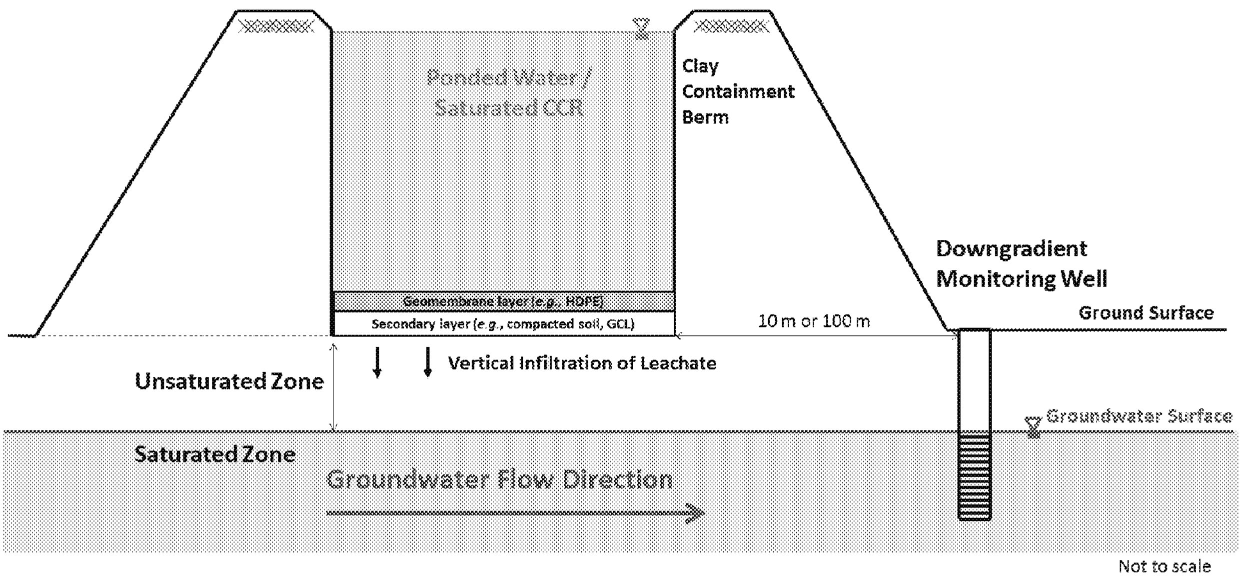

Figure 2.4 Composite-Lined Surface Impoundment Conceptual Site Model

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Table 2.1 Summary of SI Liner Scenarios Evaluated

Liner Structure Type	Scenario Evaluated
Unlined	No liner present
Engineered Clay Liner	3-ft clay, hydraulic conductivity = 10^{-7} cm/s
Natural Clay Liner	25-ft clay, hydraulic conductivity = 10^{-7} cm/s
	25-ft clay, hydraulic conductivity = 10^{-8} cm/s
	75-ft clay, hydraulic conductivity = 10^{-7} cm/s
	75-ft clay, hydraulic conductivity = 10^{-8} cm/s
	150-ft clay, hydraulic conductivity = 10^{-7} cm/s
	150-ft clay, hydraulic conductivity = 10^{-8} cm/s
	35-ft clay, variable hydraulic conductivity ranging from 5.5×10^{-9} to 2.2×10^{-8} cm/s
Composite Liner	60-mil HDPE with good-quality installation and 2-ft secondary compacted soil layer with a hydraulic conductivity of 10^{-7} cm/s
	60-mil HDPE with poor-quality installation and 2-ft secondary compacted soil layer with a hydraulic conductivity of 10^{-7} cm/s
	40-mil HDPE liner with good-quality installation (simulated by assuming twice as many defects as the default 60-mil HDPE liner) and 2-ft compacted secondary soil layer with a hydraulic conductivity of 10^{-7} cm/s
	Geomembrane consisting of a 60-mil HDPE with good-quality installation and a 0.01-m secondary GCL with a hydraulic conductivity of 10^{-9} cm/s

Notes:
GCL = Geosynthetic Clay Layer; HDPE = High-Density Polyethylene; SI = Surface Impoundment.

2.3 Hydrogeologic Environments

Infiltration from a non-intersecting CCR SI enters the natural soils in the unsaturated zone. In these strata, underlying the SI but above the water table, pore space is occupied by a mixture of air and water. The infiltrating water will flow primarily vertically downward through the pore space under the influence of gravity until it reaches the water table. Inorganic constituents transported in the dissolved phase by infiltration may sorb to the soil grains in the unsaturated zone, retarding the downward transport. After SI closure, infiltration will slow, because the overlying head in the SI will have been reduced or eliminated, and a cap will be applied to limit infiltration. However, because EPACMTP cannot simulate the transient effects to the hydraulic head that occur upon SI closure, for every unique model simulation, the hydraulic head in the SI is held constant (as a model requirement) over the entire model duration. While the source concentration term in the model is reduced to zero at SI closure, the constant hydraulic head creates a higher downward flux of water through the unsaturated zone than if dewatering and capping were simulated. As a result, mass in the unsaturated zone is transported to the saturated zone over a shorter period than if the hydraulic effects of dewatering and capping were simulated. This modeling approach was used by US EPA in its CCR Risk Assessment (US EPA, 2014).

Upon reaching the saturated zone, infiltrate will mix with groundwater and penetrate to a depth that is dependent on a variety of factors, including the vertical infiltration rate through the unsaturated zone and the horizontal velocity of groundwater in the saturated zone. In groundwater, inorganic constituents simulated with EPACMTP are assumed to migrate in a horizontal flow direction consistent with regional groundwater hydrology. Inorganic constituents may continue to sorb to the soil grains in the saturated zone, resulting in a retarded velocity relative to the migration of groundwater. During migration, the plume can spread, or disperse, longitudinally in the direction of groundwater flow (longitudinal dispersion), transverse to the direction of groundwater flow (transverse dispersion), and vertically deeper into the aquifer (vertical dispersion). Subsurface transport was simulated for migration times of up to 10,000 years to be consistent with US EPA (2014).

The hydrogeological characteristics of both the unsaturated and saturated zones were selected for each individual modeling scenario by EPACMTP using default, built-in information obtained by US EPA from regional databases. The EPACMTP databases include correlated aquifer parameters from till over sedimentary rock, sand and gravel, alluvial basins, valleys and fans, river alluvium with and without overbank deposits, till and till over outwash, unconsolidated and semi-consolidated shallow surficial aquifers, and other common hydrogeologic environments (US EPA, 2003b, p. 5-42). Specific input parameter values and input distributions are discussed in Section 3.2.

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3 Model Development

Modeling was performed to evaluate the performance of the alternative liners relative to the base case 60-mil composite liner. Modeling was conducted using a probabilistic, Monte Carlo approach, which accounts for the diversity of potential climatic, hydrogeologic, and source conditions in a statistical manner. The Monte Carlo approach described in this section was similar to the approach developed and used by US EPA in the CCR Risk Assessment (US EPA, 2014).

3.1 Model Selection

Modeling was performed using EPACMTP (US EPA, 1997). This model was developed by US EPA to simulate the fate and transport of constituents leaching from land-based waste management units through the underlying unsaturated and saturated zones. Fate and transport processes simulated by the model include advection, hydrodynamic dispersion, linear or non-linear sorption, and chain-decay reactions. The model consists of a one-dimensional analytical solution that simulates infiltration and dissolved constituent transport through the unsaturated zone, which is coupled with a three-dimensional analytical solution for transport in the saturated zone.

EPACMTP was selected for this modeling because:

- It has the capability to efficiently perform large numbers of simulations using a probabilistic, Monte Carlo approach (US EPA, 2003a);
- It was designed specifically to simulate the infiltration of constituents leaching from SIs (US EPA, 2003a); and
- It was the modeling package used by US EPA for the Agency's CCR Risk Assessment (US EPA, 2014).

EPACMTP is publicly available from US EPA.[1] The program requires control parameters, which govern the execution of the simulation, and physical parameter inputs. Control parameters include the number of Monte Carlo runs to be conducted and whether or not a liner is present at the site. Physical parameter inputs govern the physics of the simulation and include the infiltration rate and saturated zone horizontal hydraulic conductivity. Most of the physical parameter inputs can be specified as constant values, statistical distributions (uniform, normal, lognormal, etc.), empirical inputs provided by the user, derivatives from other model inputs, or values selected from regional databases. During an EPACMTP simulation, a single value is selected for each parameter from the range of potential values and used to simulate unsaturated and saturated zone transport and to predict concentrations at a downgradient receptor point.

[1] https://www.epa.gov/smm/epas-composite-model-leachate-migration-transformation-products-epacmtp

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For each liner scenario, 10,000 unique simulations were evaluated, each resulting in a single predicted maximum concentration at a downgradient point. Upon completion of all 10,000 simulations, a statistical distribution of output parameters (i.e., downgradient point concentrations) was prepared. Section 3.2 presents the parameters and distributions that were used in this evaluation. With the exception of input parameters that vary depending on which liner is being evaluated (e.g., infiltration rate), all input parameters and distributions were defined in the same manner for each scenario to allow for an appropriate "apples to apples" comparison of the different liner alternatives.

3.2 Parameter Distributions and Selections

Characteristics and parameter distributions for the evaluations presented in this white paper are described below. Because the geology underlying the SI is an important factor affecting CCR constituent transport, the modeling was performed assuming a range of potential hydrogeologic conditions. When feasible, parameter distributions similar to those used by US EPA (2014) were used for this analysis. Two inorganic constituents were evaluated – lithium and arsenic(III), the latter of which is referred to simply as "arsenic" for the remainder of this white paper. Both lithium and arsenic were identified by US EPA (2014) as potential risk-driving constituents from SIs at the 90th percentile. EPACMTP inputs (Attachment A) and the distributions for key parameters are summarized below.

- The SI was assumed to be operational for 75 years (US EPA, 2014, p. 4-7). Thus, the source duration was also assumed to be 75 years.
- Source concentrations for lithium were assumed to have a 25th percentile value of 86 µg/L, a 50th percentile value of 91 µg/L, and a 75th percentile value of 163 µg/L, consistent with data presented by US EPA in the CCR Risk Assessment (US EPA, 2014, Attachment C-2). Source concentrations for arsenic were assumed to have a 25th percentile value of 40 µg/L, a 50th percentile value of 70 µg/L, a 75th percentile value of 160 µg/L, and a 95th percentile value of 490 µg/L, consistent with data presented by US EPA in the CCR Risk Assessment (US EPA, 2014, Table 5-15).
- The SI size was fixed at approximately 100 acres (416,826 m^2).
- Infiltration rates were calculated analytically or empirically, using distributions of parameters consistent with the EPACMTP input files, and input to EPACMTP as empirical distributions (see next bullet and Section 3.2.1).
- The height of the ponded water, i.e., the hydraulic head on top of the liner system, was allowed to vary from 0 to 3 m. This range was selected due a limitation in the empirical Giroud (1997) equation that was used for this assessment to calculate infiltration though composite liners (see Section 3.2.1). The Giroud approach was used to calculate infiltration because it is a similar approach as the default approach used by EPACMTP to calculate infiltration, although it is more current and based on more-recent research than the EPACMTP approach.

  Limiting the impoundment head to 0 to 3 m also facilitated a direct comparison of the results for the various modeled scenarios. Because EPACMTP requires that the groundwater mound underneath an SI not intersect the bottom of the SI, higher impoundment heads would tend to lead to larger mounds for scenarios with higher infiltration rates, and EPACMTP would discard more mounding scenarios for some liners than for others, resulting in different sampling of the input parameters among the different liner scenarios.

  In order to evaluate the importance of hydraulic head for the infiltration rate, a sensitivity analysis was performed using the regional distribution of impoundment heads in the EPACMTP SI source module, which varied up to 19 m (see Section 4.2.1), for two liner scenarios: the base case 60-mil

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composite scenario and the 35-ft natural clay liner scenario. Because the Giroud approach is limited to calculating infiltration though composite liners with heads ranging from 0 to 3 m, two alternative approaches were used to calculate infiltration through the 60-mil composite: infiltration was calculated using EPACMTP directly and infiltration was calculated using the analytical Touze-Foltz et al. (1999) equation. The results of the sensitivity analysis are presented in Section 4.2.1, and show that use of the Giroud equation with 0 to 3 m of head did not appreciably affect results relative to the other two infiltration approaches, particularly with respect to the percentage of model simulations higher than the risk threshold for lithium and arsenic.

- The SI was assumed to be constructed on the ground surface (i.e., not incised).
- Groundwater concentrations were evaluated at two hypothetical monitoring wells (points) located 10 m and 100 m downgradient of the SI. The 10-m distance simulates a monitoring well at the waste boundary for an SI, assuming that the berm is 10 m wide at its base; the 100-m distance represents an alternative compliance monitoring distance that may be applicable for some inactive SIs that are not subject to monitoring under the CCR Rule. Monitoring points were located along the centerline of the plume, and EPACMTP adjusted the vertical position of the monitoring point to ensure that it was at the top of the plume. These settings assured that tabulated model results represent the highest calculated concentrations in groundwater because dispersion calculations within the model cause concentrations to decrease laterally from the centerline, and to decrease vertically downward from the top of the plume. In comparison, receptor well locations modeled by US EPA (2014) varied up to a distance of 1 mile from the SI, were not necessarily along the centerline of the plume, and were not necessarily within the vertical extent of the plume or at the vertical point of highest concentration. Furthermore, US EPA also considered the impact of surface water bodies positioned between the SI and the receptor well that would intercept some of the plume and reduce concentrations at the receptor well. No surface water body impacts are considered in this evaluation. These differences would result in higher predicted concentrations at the monitoring points used for this evaluation than at the receptor wells modeled by US EPA, for a case in which all other inputs would be equal.
- Aquifer hydraulic conductivities were allowed to vary based on default EPACMTP databases from 0.1 m/year to 6.6×10^5 m/year (3×10^{-7} cm/s to 2 cm/s). Hydraulic gradients were allowed to vary, based on default EPACMTP databases, from 10^{-5} to 0.1. The porosity was calculated in EPACMTP based on particle diameter and bulk density, both of which varied based on EPACMTP's default empirical distributions.
- The aquifer thickness was allowed to vary, based on built-in EPACMTP databases, from 3 to 915 m.
- Unsaturated zone parameters, including the unsaturated zone hydraulic conductivity, van Genuchten equation parameters, and water content, were allowed to vary based on built-in EPACMTP databases.

Lithium was modeled as a conservative tracer, i.e., no retardation or sorption was specified in either the unsaturated or saturated zones, consistent with US EPA (2014). Retardation coefficients were calculated for arsenic to simulate the reduced transport velocity due to sorption. In the unsaturated zone, a distribution of arsenic soil-water partition coefficient (K_d) values consistent with US EPA (2014) was specified, and EPACMTP calculated the retardation factor (R).[2] For the saturated zone, the retardation factor was a direct input to EPACMTP.

[2] Note that neither saturated nor unsaturated retardation was considered within the liner layers, including the full thickness of the natural clay liner, as explained in Section 3.3.

The saturated zone retardation factor was calculated using Equation 1.

$$R = 1 + \frac{\rho_b K_d}{\theta} \qquad \text{Eq. 1}$$

where:

R = Saturated zone retardation factor
ρ_b = Soil bulk density
K_d = Soil-water partition coefficient
θ = Effective porosity

A soil bulk density (ρ_b) of 1.65 g/cm^3 was used, consistent with EPACMTP's default specification. An effective porosity (θ) of 0.3 was similarly used. A distribution of arsenic soil-water partition coefficient values based on US EPA (2014) was used. The soil-water partition coefficients and the calculated retardation factors are presented in Table 3.1.

Table 3.1 Arsenic Soil-Water Partition Coefficients and Retardation Factors

Percentile	5%	10%	25%	50%	75%	90%	95%
Saturated K_d (cm^3/g)[a]	0.11	0.18	0.31	0.87	1.4	1.8	2.2
Saturated R (unitless)	1.6	2.0	2.7	5.8	8.7	11	13

Notes:
K_d = Soil-Water Partition Coefficient; R = Saturated Zone Retardation Factor
(a) Source: US EPA (2014).

3.2.1 Infiltration Rate Calculations

Infiltration rates were calculated analytically or empirically and used as an input to EPACMTP. First, EPACMTP was run to generate sets of input parameters. For each of the 10,000 combinations of input parameters generated by EPACMTP, infiltration rates were calculated analytically and empirically, as described below, resulting in 10,000 infiltration rates. Percentile statistics (e.g., 25th percentile, 50th percentile) of the 10,000 calculated infiltration rates were generated (Table 3.2). The percentiles were input to EPACMTP as an empirical distribution for the modeled infiltration rate parameter and EPACMTP was run again to produce the final model results.

For the unlined, engineered clay, and natural clay scenarios, infiltration from the SI, through the liner, and into the underlying soils was estimated using Darcy's Law. In circumstances in which there is no hydraulic pressure underlying a liner causing resistance to downward seepage, which is consistent with the CSM developed for the modeling performed in this research, the gravity-driven specific discharge in a saturated clay liner can be quantified by the following equation.

$$q = k_s\left(\frac{h}{t_s} + 1\right) \qquad \text{Eq. 2}$$

where:

q = Specific discharge, which is the unit flow rate per area across the liner
k_s = Saturated hydraulic conductivity of the liner

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h = Pressure head of the liquid on top of the liner (limited to 0 to 3 m to maintain consistency with the composite liner infiltration calculations)
t_s = Thickness of the liner

For composite liner scenarios, infiltration was calculated using the Giroud approach (Giroud, 1997), which is a semi-empirical method of calculating flow through a defect in a composite liner (Equation 3).[3] The Giroud approach is similar to, but more current than, the default approach used in EPACMTP to calculate infiltration through a composite liner (US EPA, 2003a, p. 2-30).

$$Q_d = 0.976\, C_{qo}[1 + 0.1(h/t_s)^{0.95}](2r_d)^{0.2}h^{0.9}{k_s}^{0.74} \qquad \text{Eq. 3}$$

where:

Q_d = Flow rate through a single defect
h = Pressure head of the liquid on top of the liner
t_s = Thickness of the liner
r_d = Radius of circular defect
k_s = Saturated hydraulic conductivity

Here, C_{qo} is a unitless coefficient that varies in value depending on whether contact conditions are good (C_{qo} = 0.21) or poor (C_{qo} = 1.15). Good contact is when the GM has few wrinkles and the clay layer has a smooth surface. Poor contact is when the GM has many wrinkles and the clay layer has a rough surface. The Giroud formula is semi-empirical and only valid for head values (h) ranging from 0 to 3 m and when inputs are expressed in the following units: Q_d (m^3/s), i_s (unitless), r_d (m), h (m), and k_s (m/s).

Specific parameters that were used to calculate infiltration through a composite liner are summarized below.

- The defect radius was assumed to be 2.8 mm (equivalent to an area of 6 mm^2; US EPA, 2003a, p. 2-30).
- For the 60-mil HDPE scenario, the defect density was assumed to follow an empirical distribution ranging from 0-12.5 defects/hectare based on US EPA (2003b, p. 2-23). To simulate the effects of a thinner composite, a defect density of double the default defect density was evaluated. This is a conservative assumption, as discussed in Section 2.2.
- Model simulations were evaluated for both a good and poor contact coefficient (C_{qo} = 0.21 and 1.15; Giroud, 1997).

For all of the infiltration calculations, the distributions of the input parameters to be used in the equations were created using EPACMTP, and then the table of 10,000 input values was used to calculate the infiltration rate using Equation 2 or 3. Statistics summarizing the calculated infiltration rates for each liner scenario are provided in Table 3.2.

[3] Sensitivity analyses were also conducted using hydraulic head inputs varying from 0 to 19 m (Section 4.2). This range of hydraulic head inputs exceeds the range of heads for which the Giroud (1997) approach is valid.

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Table 3.2 Calculated Infiltration Rates for Each Liner Scenario

Liner Scenario	Calculated Infiltration Rate (m/year)		
	25th Percentile	50th Percentile	75th Percentile
Unlined	0.4	2	20
Engineered Clay Liner	0.06	0.08	0.1
Natural Clay Liner: 25 ft, $K = 10^{-7}$ cm/s	0.03	0.04	0.04
Natural Clay Liner: 25 ft, $K = 10^{-8}$ cm/s	0.003	0.004	0.004
Natural Clay Liner: 75 ft, $K = 10^{-7}$ cm/s	0.03	0.03	0.03
Natural Clay Liner: 75 ft, $K = 10^{-8}$ cm/s	0.003	0.003	0.003
Natural Clay Liner: 150 ft, $K = 10^{-7}$ cm/s	0.03	0.03	0.03
Natural Clay Liner: 150 ft, $K = 10^{-8}$ cm/s	0.003	0.003	0.003
Natural Clay Liner: 35 ft, $K = 5.5 \times 10^{-9}$ to 2.2×10^{-8} cm/s	0.003	0.005	0.006
Baseline (60-mil) Composite Liner	0	6E-5	2E-4
Baseline Composite Liner with Poor-Quality Installation	0	3E-4	0.001
40-mil Composite Liner (Defect Density 2X Base Composite)	0	7E-5	3E-4
60-mil Composite Liner with GCL	0	1E-5	7E-5

Notes:
GCL = Geosynthetic Clay Layer; K = Hydraulic Conductivity.

The highest infiltration rates are associated with the unlined scenario. The lack of a liner means that the infiltration rate is governed primarily by the hydraulic conductivity of the underlying soils. Scenarios that included liners all had median infiltration rates more than an order of magnitude lower than the unlined scenario. Engineered clays had the highest infiltration rate among the liner scenarios. Infiltration rates for the natural clays are directly controlled by the hydraulic conductivity, such that the infiltration rates for a clay with a hydraulic conductivity of 10^{-8} cm/s are an order of magnitude less than those with a hydraulic conductivity of 10^{-7} cm/s. The composite liner scenarios all had zero infiltration below the 50th percentile; this is because nearly 50% of composite liner systems have no defects based on the empirical distribution of defects used by EPACMTP (US EPA, 2003b, p. 2-23).

3.3 Modeling Limitations

Any groundwater modeling exercise is limited by the assumptions required to reduce highly complex natural systems to a finite set of modeling parameters. In addition to the broadly applicable modeling limitations of EPACMTP described by US EPA (2003a,b), the limitations associated with applications presented in this white paper include:

- EPACMTP assumes a constant hydraulic head and infiltration rate throughout the entire 10,000-year simulation period, which results in overcalculation of hydraulic flushing though the liner and the unsaturated zone. This is a conservative assumption for natural clay layers, for which the increased flushing may result in liner penetration for circumstances in which penetration may not otherwise occur within the simulation period.
- EPACMTP assumes a steady-state (i.e., not changing over time) infiltration rate through the liner and into the top of the unsaturated zone starting from the beginning of each simulation, i.e., the liner is initially fully saturated. This also means that the constituent mass loading through the liner into the underlying unsaturated zone is constant over the entire 75-year SI operational period. For thick natural clay liners such as those modeled here, the time for leachate to migrate through the natural clay may be longer than the operational period of the SI. The result of this model limitation

is that the chemical mass for the natural clay liner scenario enters the aquifer more quickly and over a shorter period than if modeled infiltration rates decreased when the impoundment was removed or capped. Because a mixing calculation is used to determine concentration in the saturated zone, inputting the chemical mass over a shorter period can result in less mixing and higher predicted concentrations in the saturated zone than inputting the same mass over a longer period. The effect of starting with an initially unsaturated clay liner, rather than an initially saturated clay liner (as described above), is addressed in the sensitivity analyses (Section 4.2.2).

- The time of migration through the natural clay liner is not evaluated in EPACMTP, resulting in a significant underestimation of travel times.[4] This limitation is addressed in the sensitivity analyses (Section 4.2.2).
- For liners that have unsaturated conditions over part of their thickness, as is possible with thick natural clay liners if the water table is deep, the rate of infiltration will be reduced in the unsaturated portion of the liner. This is because, in a multi-fluid system, fluids (in this case, air and water) compete for pore space, resulting in reduced mobility for all fluids (Cohen and Mercer, 1993). This may result in the model overestimating the mass flux entering the aquifer, resulting in higher maximum predicted concentrations at the downgradient receptor wells, because less mixing with the regional aquifer will occur.
- The mass of contaminants sorbed to the thick natural clay liner is not considered as part of the analysis. This is only relevant for arsenic, because lithium was modeled as a conservative tracer. Thus, the migration rate of arsenic through the liner is overpredicted by the model. This limitation can result in arsenic mass entering the system over a shorter period than if retardation in the liner was considered, resulting in less mixing and higher predicted maximum concentrations than if retardation in the liner was considered. This limitation is addressed in the sensitivity analyses (Section 4.2.2).
- Simulations were run to identify the peak downgradient concentration over an exposure period of 10,000 years, to be consistent with US EPA (2014). Typical human health risk assessments assess exposures over a 30-year period, and 10,000 years is a much longer period over which to evaluate migration and assess potential risk.
- Assuming constant liner conditions over a 10,000-year simulation period is uncertain. For example, current research indicates that geomembrane composite liners have a finite life (Gulec et al., 2004; Andrews and Loellen, 2008; Geosynthetic Institute, 2011; Tian et al., 2017) that is less than the model simulation period.[5]
- The SI was simulated as a constant, infinite source of mass. In reality, the mass of inorganic constituents that leach from an SI is finite and can be depleted over time due to leaching. US EPA assumed that the constant-strength source assumption is valid over the simulated operational lifetime of the impoundment. It is uncertain if the 75-year SI source term used by US EPA and in this evaluation is conservative or not conservative. EPRI is currently performing research to better understand leaching duration from CCR landfills and SIs.
- Linear sorption coefficients were assumed for arsenic. The mass transfer from the sorbed to the dissolved phase, and the kinetics associated with this, were not simulated.
- All of the SIs modeled were assumed to have non-intersecting groundwater conditions, i.e., the SIs have an incised depth less than or equal to the pre-development depth to groundwater, consistent

[4] Travel time to the monitoring point as calculated by the model, which ignores travel time through the liner, will be shorter than if the model accounted for travel time through the liner.
[5] Benson (2019) notes that after a unit reaches hydrostatic equilibrium, the cap becomes the controlling mechanism for infiltration of water into, and out of, a waste management unit. This assumes no active leachate collection and that the cap allows less vertical flow than the liner.

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with US EPA (2014, p. 4-9). Furthermore, due to modeling restrictions in the US EPA software package, SI characteristics were selected such that the size of the groundwater mound produced by the SI does not intersect the bottom of the SI (US EPA, 2003a, p. 2-39 to 2-41). Thus, an unsaturated zone was always used in the modeling, implying no hydraulic pressure on the bottom of the SI. The presence of groundwater immediately under the SI would reduce the hydraulic gradient and thus reduce the infiltration rate through the soil liners. The effect of hydraulic pressure on the bottom of a natural clay liner was evaluated in the sensitivity analyses (see Section 4.2.3).

- Reactive transport (i.e., the transition between the two arsenic species and the formation of different soluble and insoluble metal complexes) was not simulated. Similarly, the redox processes affecting arsenic transport were not explicitly simulated. The presence of carbonate minerals, iron oxides, variable pH, and other conditions controlling the geochemical fate and transport of arsenic may affect the arsenic concentrations in a natural system in ways not simulated by the modeling.

3.4 Model Assumptions

Table 3.3 summarizes the assumptions used in this modeling that varied from those used by US EPA in the Agency's CCR Risk Assessment (US EPA, 2014). The overall effect of these differences in assumptions is that the model-calculated maximum constituent concentrations, and the resulting estimates of risk to HHE, calculated by this model are expected to be greater than the concentrations and risk that would be derived using the assumptions of US EPA (2014). Furthermore, it is important to recognize that this modeling was performed to compare relative liner performance, and the results are not applicable to any specific site due to the limitations and conservative assumptions used in the modeling.

Table 3.3 Summary of Modeling Assumptions That Differed from US EPA's CCR Risk Assessment

Modeling Limitations	Potential Impact Relative to US EPA's CCR Risk Assessment
SI area was fixed at approximately 100 acres.	Negligible. Monitoring points for this evaluation were limited to the plume centerline at downgradient distances of 10 m and 100 m – an assumption that limits the importance of the SI area. The SI size is a more important parameter when monitoring points are allowed to vary laterally off of the plume centerline.
Results were assessed for monitoring points 10 m and 100 m downgradient of the SI, along the plume centerline, with no surface water between the source and the monitoring point.	Conservative. The model-predicted constituent concentrations are higher than if the locations varied over greater distances or were not located on the plume centerline (as was done in the CCR Risk Assessment)
Results assessed at top of the plume.	Conservative. The model-predicted concentrations are higher than if the monitoring depths were allowed to vary vertically within the plume (as was done in the CCR Risk Assessment)
Infiltration rates were calculated analytically, using distributions of parameters consistent with the EPACMTP input files, and input to EPACMTP as empirical distributions.	Negligible. The approach used to calculate infiltration is similar to the approach used by EPACMTP. The impact of this assumption was evaluated in the sensitivity analysis (Section 4.2.1).
Head on liner was limited to a maximum value of 3 m.	Negligible. See the sensitivity analysis (Section 4.2.1)

Notes:
CCR = Coal Combustion Residual; EPACMTP = US EPA Composite Model for Leachate Migration and Transformation Products; SI = Surface Impoundment; US EPA = United States Environmental Protection Agency.
CCR Risk Assessment = US EPA (2014).

4 Modeling Results

Based on the CSMs presented in Section 2 and the modeling approach described in Section 3, the performance of the alternative liners, relative to the base case, was evaluated using the probabilistic approach available in EPACMTP. Model results are presented below.

4.1 Model-Predicted Groundwater Concentrations

The relative performance of each liner scenario was evaluated at two downgradient locations over a period of 10,000 years. For each constituent and for each liner scenario, 10,000 unique simulations in EPACMTP were evaluated. Information for obtaining EPACMTP model input files is provided in Appendix A. EPACMTP also produces output files containing the parameter values for each of the 10,000 simulations for many of the modeled parameters. Information for obtaining output results is provided in Appendix B.

A probabilistic presentation of relative maximum predicted lithium concentrations in groundwater, at 10 m and 100 m downgradient of the SI, is provided in Figures 4.1a and 4.1b. A probabilistic presentation of relative maximum predicted arsenic concentrations in groundwater, at 10 m and 100 m downgradient of the SI, is provided in Figures 4.2a and 4.2b. The figures also show the US EPA Regional Screening Level (RSL; US EPA, 2018a) for lithium or the US EPA Maximum Contaminant Level (MCL; US EPA, 2018b) for arsenic, so that relative performance can be discussed in terms of HHE. Note that these model results are conservative, are only intended for evaluation of relative liner performance, and do not necessarily indicate actual or potential adverse impacts to HHE.

Specific conclusions based on the modeling results are summarized below. The conclusions focus on the results of probability distributions between the 10^{th} and 90^{th} percentiles as most representative of overall scenario behavior. This is consistent with US EPA's guidance for conducting probabilistic risk assessments and evaluating probabilistic data distributions (US EPA, 2001). This guidance states that:

> *[T]he extreme percentiles ("tails") of an input distribution are understandably the most uncertain part of a [distribution]... since the number of data values in these ranges are less abundant than in the center of the range. This uncertainty in the tails of the input distributions leads in turn to greater uncertainty in the tails of the calculated exposure or risk distribution, and the magnitude of this uncertainty increases rapidly at the very high percentiles. In many cases, estimates at the extreme tails, such as the 99.9^{th} percentile, may be neither accurate nor plausible. For that reason, great care should be taken when evaluating an RME [reasonable maximum exposure] risk in the upper percentiles of the risk range.* (US EPA, 2001, p. 7-11)

- All of the composite liners simulated in this evaluation performed similarly, with low (<0.13%) percentages of scenarios having maximum model-predicted concentrations higher than the lithium RSL or arsenic MCL. Even composite liners with a high defect density, representative of a thin (40 mil), non-federally compliant GM, achieved a level of protection comparable to the baseline composite scenario.

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- The unlined SI scenario had the highest predicted downgradient concentrations, with more than three-quarters of the simulations having a concentration higher than the reference RSL or MCL. As previously noted, this scenario is not considered a liner alternative; rather, it provides insight into the results that this model will predict for an uncontrolled scenario.
- The engineered clay liner scenario was more similar to the uncontrolled scenario than to the base case composite liner scenario for both arsenic and lithium. At 10 m downgradient of an SI with an engineered clay liner, modeled groundwater concentrations exceeded the RSL for lithium in 72% of simulations and exceeded the MCL for arsenic in 84% of simulations. These percentages are higher than the percentage of RSL/MCL exceedances calculated by US EPA (2014), which is consistent with the different assumptions used in the two evaluations (i.e., monitoring point locations).
- Model results for the low-hydraulic-conductivity (i.e., 10^{-8} cm/s) natural clay liners were more similar to the base case composite liner scenario than to the unlined scenario for both lithium and arsenic. Model-predicted maximum lithium concentrations for the natural liner scenario with a hydraulic conductivity of 10^{-8} cm/s were less than the lithium RSL in 93 to 94% of the simulations (depending on liner thickness) 10 m downgradient and 98 to 99% of the simulations 100 m downgradient. Model-predicted maximum arsenic concentrations were less than the arsenic MCL in 86 to 88% of simulations 10 m downgradient and 94 to 95% of simulations 100 m downgradient.
- The specific natural clay liner simulations that were evaluated with a thickness of 35 ft and hydraulic conductivity values ranging from 5.5×10^{-9} to 2.2×10^{-8} cm/s, representative of the measured values in the case study site (Section 5), were similar to the low-hydraulic-conductivity scenarios, with model-predicted maximum lithium concentrations less than the lithium RSL in 88% of the simulations at a distance of 10 m from the source and in 96% of the simulations at a distance of 100 m from the source. Model-predicted maximum arsenic concentrations were less than the arsenic MCL in 81% of simulations at a distance of 10 m from the source and in 90% of simulations at a distance of 100 m from the source.
- The hydraulic conductivity of the natural clays is a sensitive parameter in these evaluations. As the hydraulic conductivity of a natural clay layer is decreased, the distribution becomes more similar to the base case. Conversely, the natural clay liners with a modeled hydraulic conductivity of 10^{-7} cm/s yielded concentration distributions between the 10^{-8} cm/s natural clay scenario and the engineered clay liner. The modeled peak concentrations of lithium for the natural clay scenario at 10^{-7} cm/s was higher than the lithium RSL in more than 40% of simulations, while at 10^{-8} cm/s, it was higher than the lithium RSL in fewer than 7% of simulations. For arsenic, the 10^{-7} cm/s natural clay liner scenario resulted in modeled peak concentrations higher than the arsenic MCL in more than 50% of simulations, while the 10^{-8} cm/s scenario resulted in maximum concentrations higher than the MCL in less than 14% of simulations. Conversely, the model had low sensitivity to the thickness of the natural clay liner, within the range (25 to 150 ft) modeled.

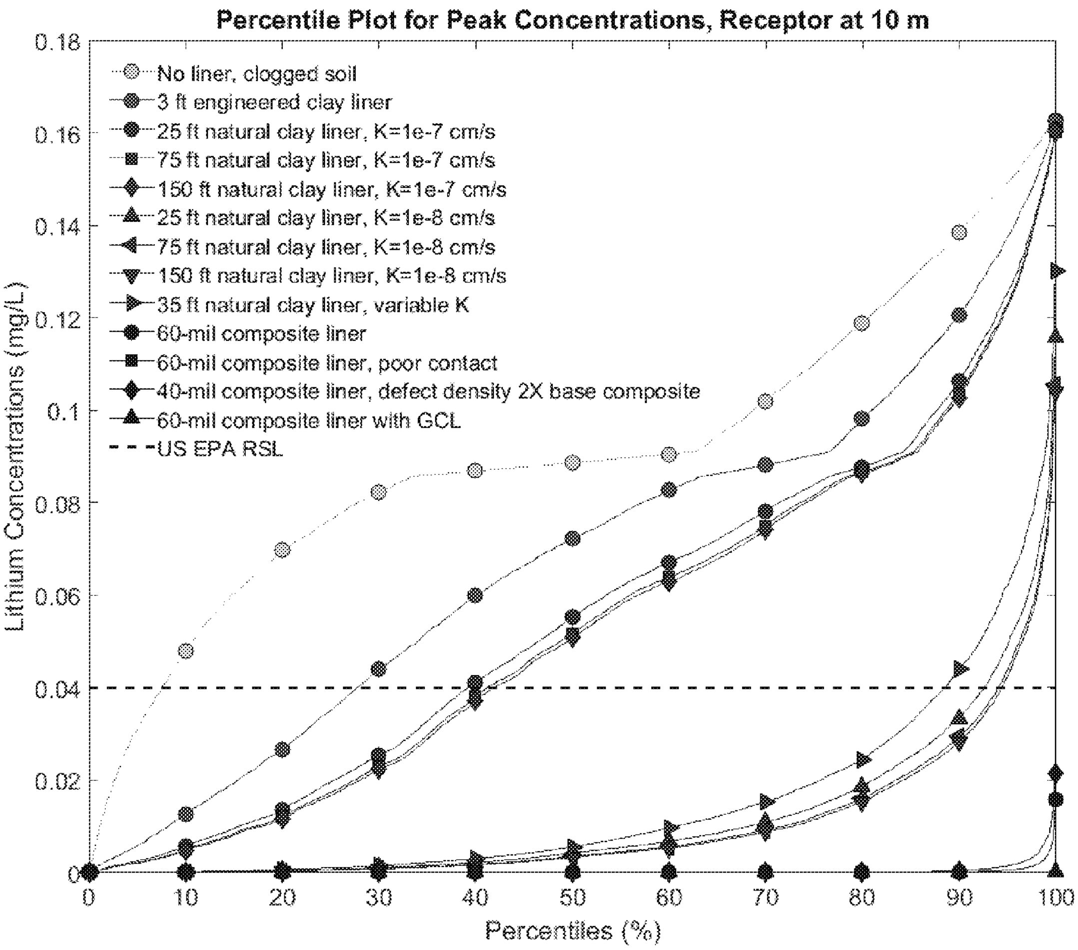

Figure 4.1a Probabilistic Relative Distribution of Lithium Concentrations in Groundwater at a Receptor Well 10 m Downgradient

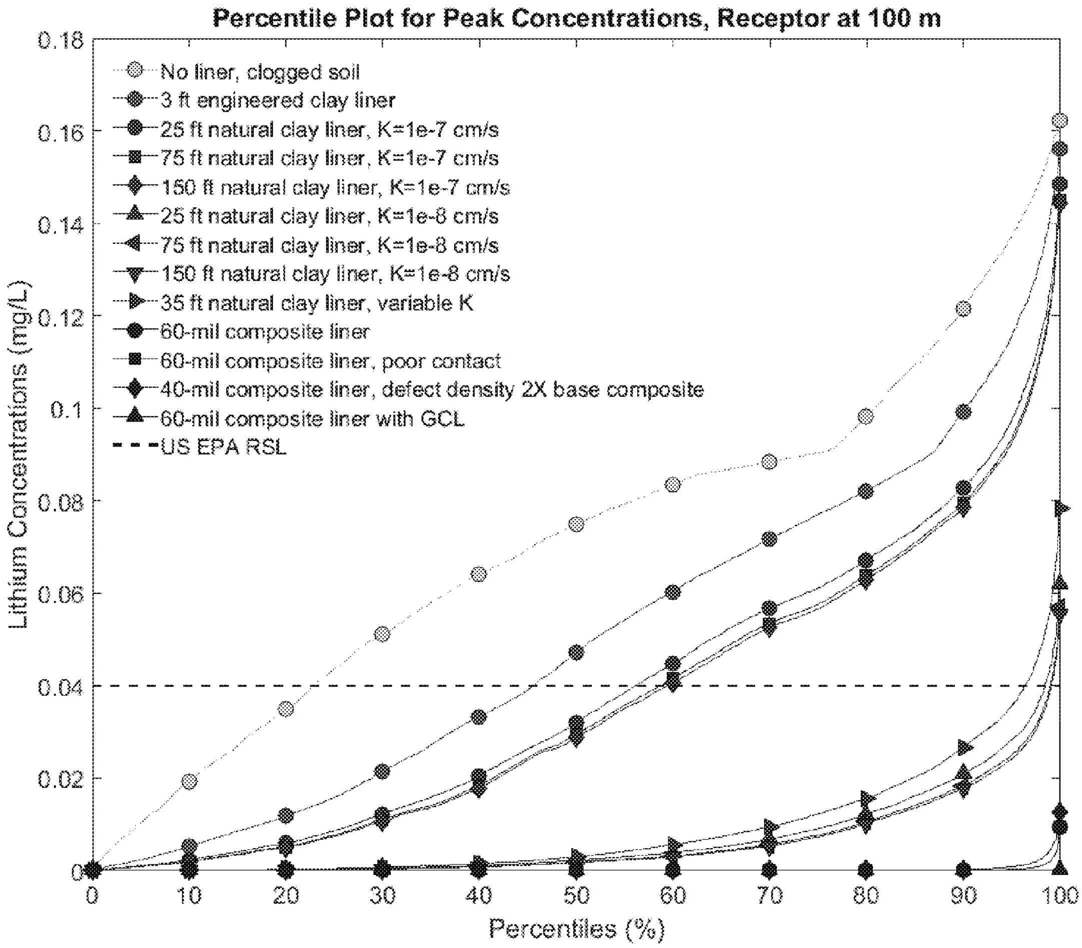

Figure 4.1b Probabilistic Relative Distribution of Lithium Concentrations in Groundwater at a Receptor Well 100 m Downgradient

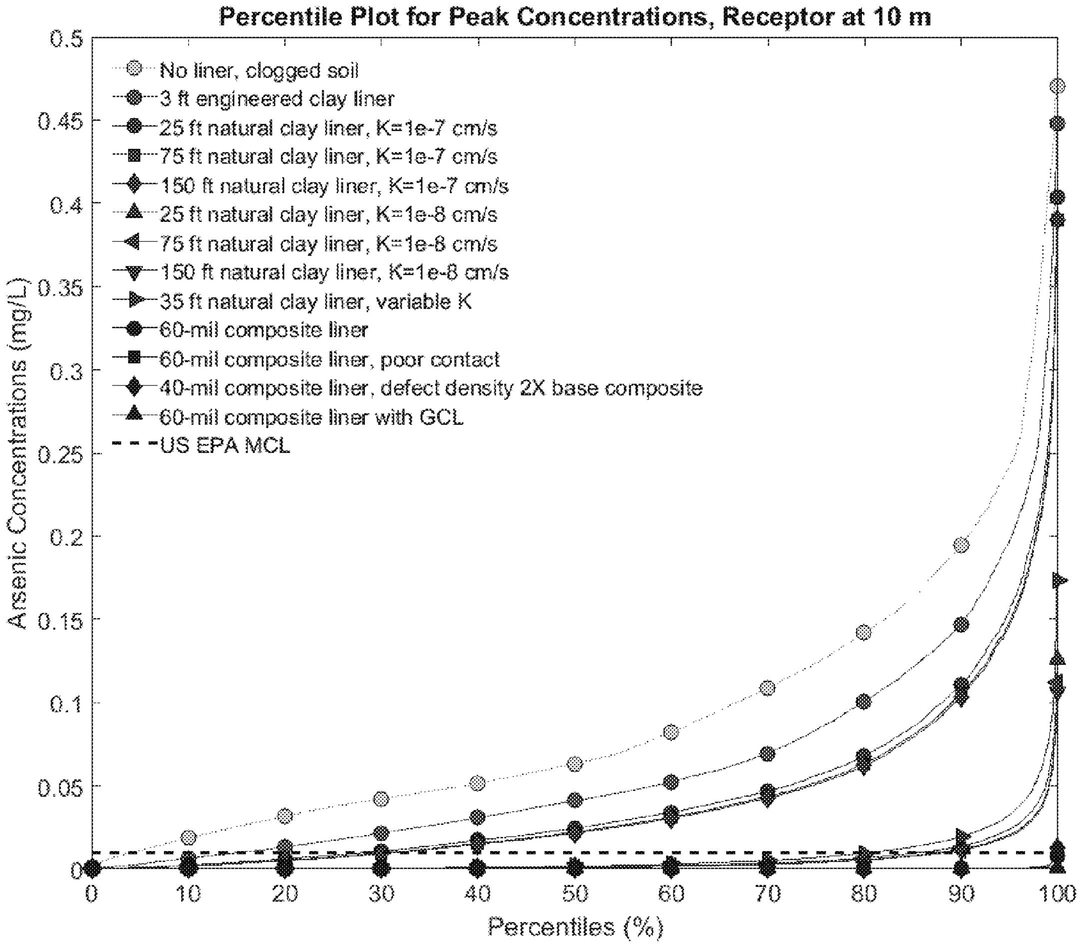

Figure 4.2a Probabilistic Relative Distribution of Arsenic Concentrations in Groundwater at a Receptor Well 10 m Downgradient

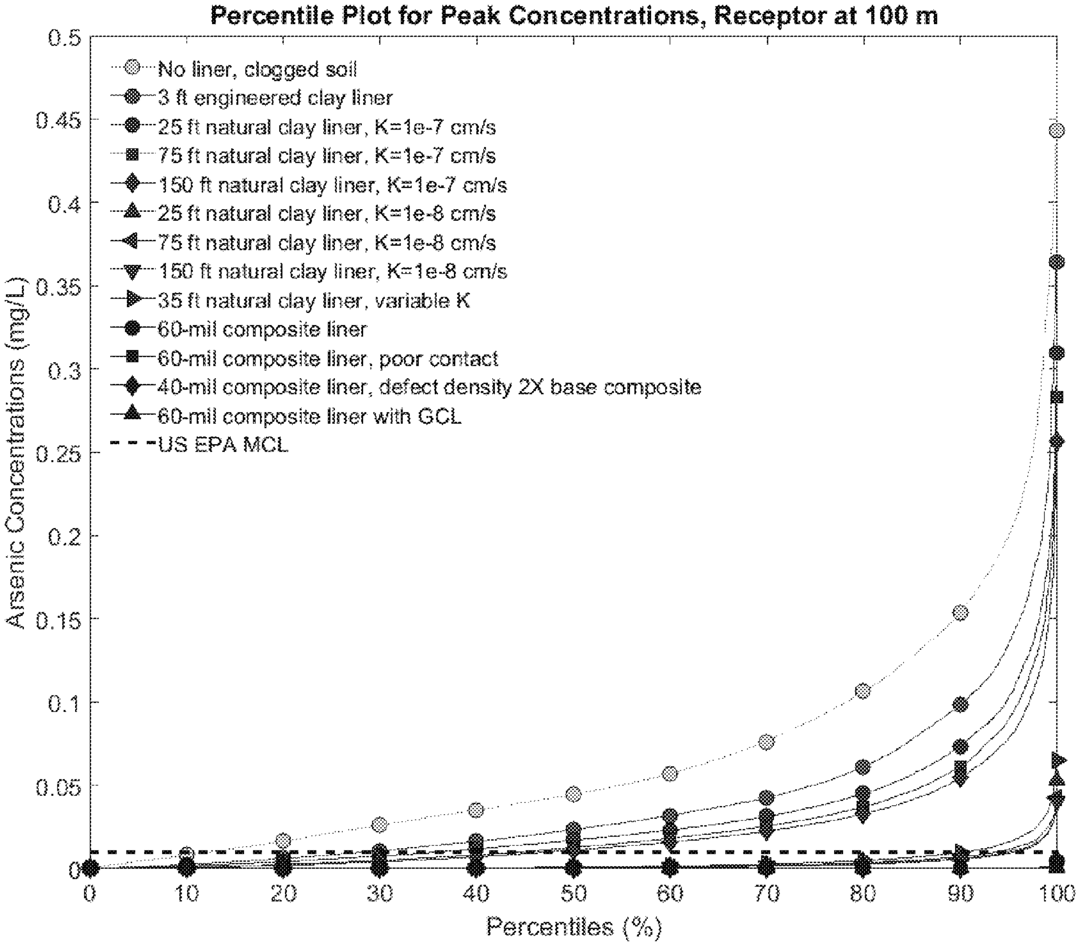

Figure 4.2b Probabilistic Relative Distribution of Arsenic Concentrations in Groundwater at a Receptor Well 100 m Downgradient

4.2 Sensitivity Analyses

Sensitivity analyses were performed to test several modeling assumptions and restrictions, summarized below.

1. The restriction that SI hydraulic heads be limited to 3 m or less.
2. The impact of modeling natural clays as an EPACMTP-defined "liner" (Figure 2.3). This modeling construct was intended to ensure consistency across the different modeling scenarios; however, by simulating natural clays as an EPACMTP-defined "liner," rather than as part of the unsaturated zone, several conservative limitations were introduced into the modeling, specifically:
   a. Constituent migration through the natural clays was not considered. A steady-state contaminant mass loading from the bottom of the natural clay liner over the 75-year SI operational period was assumed; and
   b. Sorption of arsenic was not considered during migration through the natural clay liner. Lithium was modeled as a conservative tracer and is not subject to sorption.
3. Groundwater underlying the natural clay liner was required to be unconfined, with no net positive pressure head at the base of the natural clay liner. This is due to the limitation, required by EPACMTP, that groundwater cannot intersect the SI liner.

4.2.1 Sensitivity Analysis – Impact of Hydraulic Head in SI

A sensitivity analysis was performed to test the parameter input restriction that the hydraulic head present in the SI be less than or equal to 3 m, which was a validity requirement for the Giroud (1997) equation (Section 3.2.1). The sensitivity analysis was conducted using two scenarios: the 35-ft natural clay liner with hydraulic conductivity varying from 5.5×10^{-9} to 2.2×10^{-8} cm/s, and the baseline 60-mil composite liner. For these sensitivity analysis scenarios, an infiltration rate distribution was calculated using the regional distribution data of SI hydraulic heads built into EPACMTP, which include SI hydraulic head values up to 19 m. Although the Giroud approach (Giroud, 1997) was selected for the primary analysis, because it is similar to, but more current than, the default approach used in EPACMTP to calculate infiltration through a composite liner (US EPA, 2003a, p. 2-30), it is only valid for impoundment heads up to 3 m. For this sensitivity analysis, two alternative approaches were used to calculate infiltration: infiltration was calculated using EPACMTP directly and infiltration was calculated using the analytical Touze-Foltz et al. (1999) equation. As described in Section 3.2.1, EPACMTP was run to generate sets of input parameters, infiltration rates were calculated for each of the 10,000 combinations of input parameters, percentile statistics were generated, the percentiles were input to EPACMTP as an empirical distribution for the modeled infiltration rate parameter, and EPACMTP was run again to produce the final sensitivity analysis results.

Figures 4.3 and 4.4 plot the relative results for lithium and arsenic at a receptor well located 10 m downgradient of the SI for the two sensitivity tests. Overall, the evaluation indicates that the model results are not sensitive to hydraulic heads in the impoundment, and the use of the Giroud (1997) equation with a head distribution from 0 to 3 m does not impart a significant limitation on the evaluation.

- Differences between the composite liner scenarios were negligible.
- For the natural clay liner scenario, the approach using the Touze-Foltz et al. (1999) equation with up to 19 m of head yielded maximum concentrations that were slightly higher than the Giroud

equation with up to 3 m head, although the overall effect was negligible when compared to the differences between liner scenarios (Figures 4.1a-4.2b).

- For the natural clay liner scenario, the comparison between the EPACMTP approach and Giroud approach differs by analyte:
  - For lithium, the EPACMTP approach resulted in a concentration curve that was lower than the Giroud approach up to about the 90th percentile, and then higher than the Giroud approach.
  - For arsenic, the EPACMTP approach resulted in a concentration curve that was similar to the Giroud approach until about the 90th percentile, and then higher.
  - In both cases, the percentile at which model-calculated concentrations begin to exceed the RSL for lithium and the MCL for arsenic is similar for the EPACMTP and Giroud approaches.

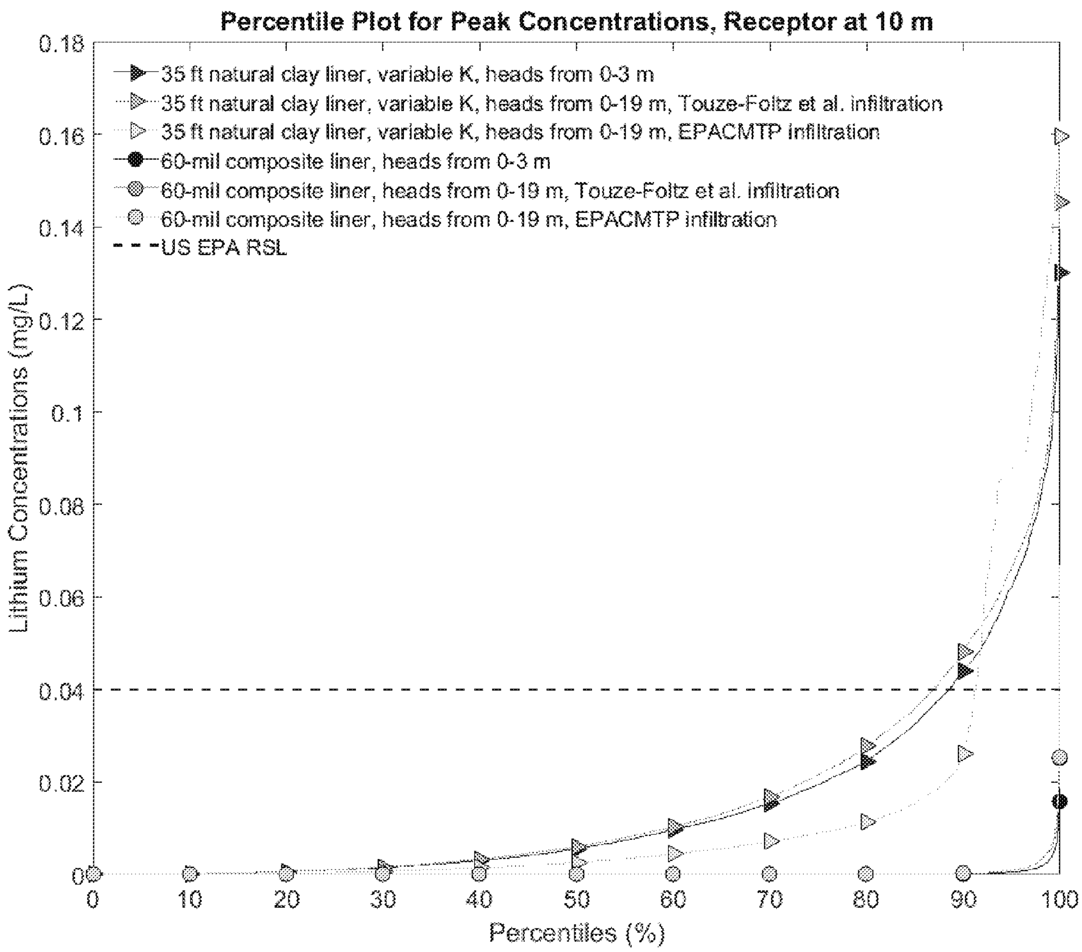

Figure 4.3 Sensitivity Analysis of Results to Impoundment Hydraulic Head – Lithium

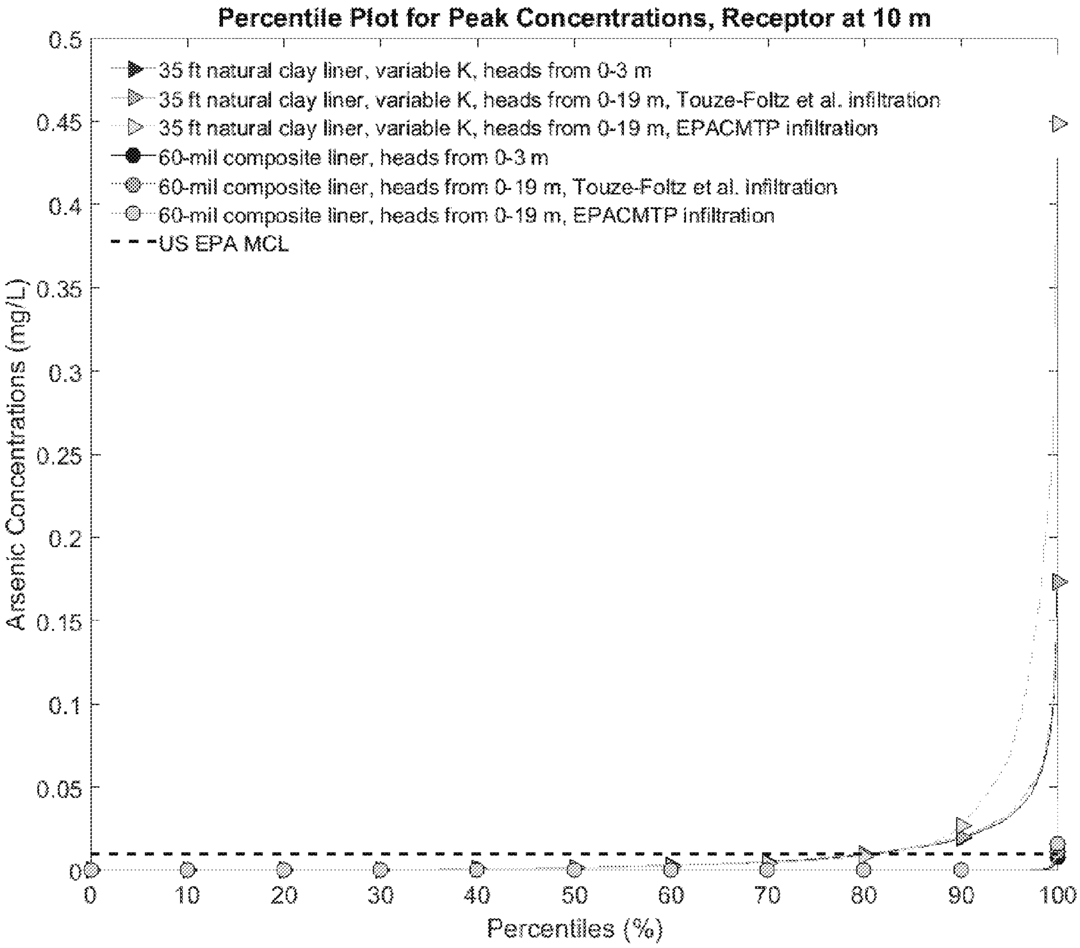

Figure 4.4 Sensitivity Analysis of Results to Impoundment Hydraulic Head – Arsenic

4.2.2 Sensitivity Analysis – Impact of Natural Clay Liner Modeling Approach

To facilitate comparison among liner scenarios, the natural clay liners were modeled assuming that the clays were part of the liner system (Figure 2.3; Section 4.1). This approach ensured that a consistent set of model input parameters were used for each liner scenario. However, this approach created several limitations; specifically, the model failed to simulate constituent travel through the natural clay liner and the model failed to account for the reduced migration rate of arsenic through the natural clay layer due to sorption. To assess the impact of modeling the natural clay liner as an EPACMTP model-specified liner, a new scenario was simulated in which the natural clay liner is modeled as the unsaturated zone (Figure 4.5). For this scenario, there is no additional unsaturated zone underlying the natural clay layer. Additionally, infiltration from the SI into the subsurface begins at the top of the natural clay layer immediately under the impounded CCR. Contaminant transport through the natural clay liner is thus simulated explicitly by EPACMTP.

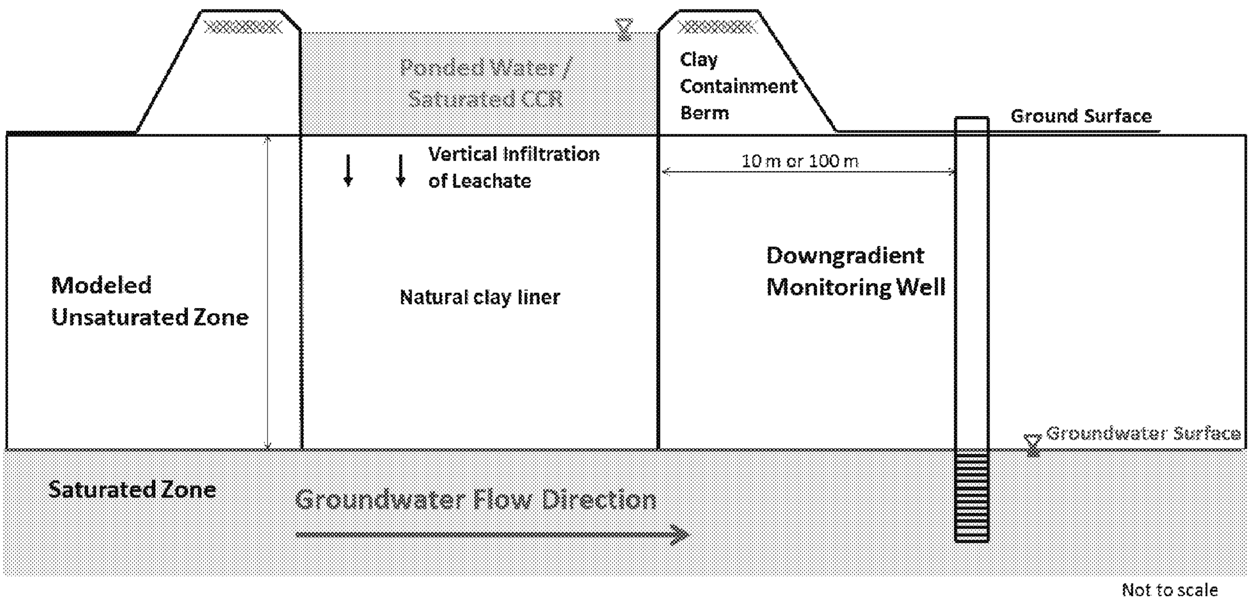

Figure 4.5 Sensitivity Analysis – Scenario in Which Natural Clay Layer Is Modeled as the Unsaturated Zone

Model sensitivity was tested for two scenarios: (1) the 35-ft natural clay liner with variable hydraulic conductivity ranging from 5.5×10^{-9} to 2.2×10^{-8} cm/s, and (2) the 150-ft natural clay liner with a hydraulic conductivity of 10^{-7} cm/s. The specification of fixed unsaturated zone characteristics (fixed saturated hydraulic conductivity and thickness) in EPACMTP precludes the use of EPACMTP regional databases to define other hydrogeologic characteristics. However, in order to maintain consistency between model runs, model input parameters that were previously sourced from regional databases, other than unsaturated zone thickness and conductivity, were defined empirically based on the statistical distributions (0th, 10th, 25th, 50th, 75th, 90th, and 100th percentiles) of the input parameters generated by EPACMTP for the prior model runs. The infiltration rates used for these analyses are the same as for the corresponding scenarios presented in Table 3.2.[6]

[6] Infiltration rates in all of the EPACMTP models are steady-state (i.e., do not change over time). The use of infiltration rates calculated using Darcy's Law implies a fully saturated natural clay liner; in reality, a natural clay liner may be unsaturated or partially saturated at the start of SI operation.

Figures 4.6 and 4.7 show the results of the sensitivity analyses using the updated natural clay modeling scenario compared with the prior results presented in Section 4.1. Addressing the prior model limitations relating to constituent migration through the liner and sorption resulted in lower modeled maximum downgradient concentrations for both the 35-ft and 150-ft natural clay layers.

Simulating contaminant transport through the natural clay liner explicitly allows for simulation of contaminant dispersion, which reduces the peak mass flux into the underlying aquifer at any given time and thus reduces downgradient concentrations. Figure 4.8 shows a plot comparing two lithium breakthrough curves at the water table for individual model runs (one of the 10,000 runs per simulation) from modeling the natural clays as an EPACMTP-defined liner and from modeling the natural clays as an unsaturated zone. The individual runs were selected to have similar infiltration rates (within 10% of each other) and source concentrations (on the order of 0.1 mg/L). The area under the curves shown in Figure 4.8, representing the mass per area entering the saturated zone under the SI, is similar between the two scenarios (within 10% of each other). However, the clay modeled as an unsaturated zone has a much lower peak concentration entering the saturated zone due to the contaminant dispersion that occurs during transport through the natural clay unsaturated zone.

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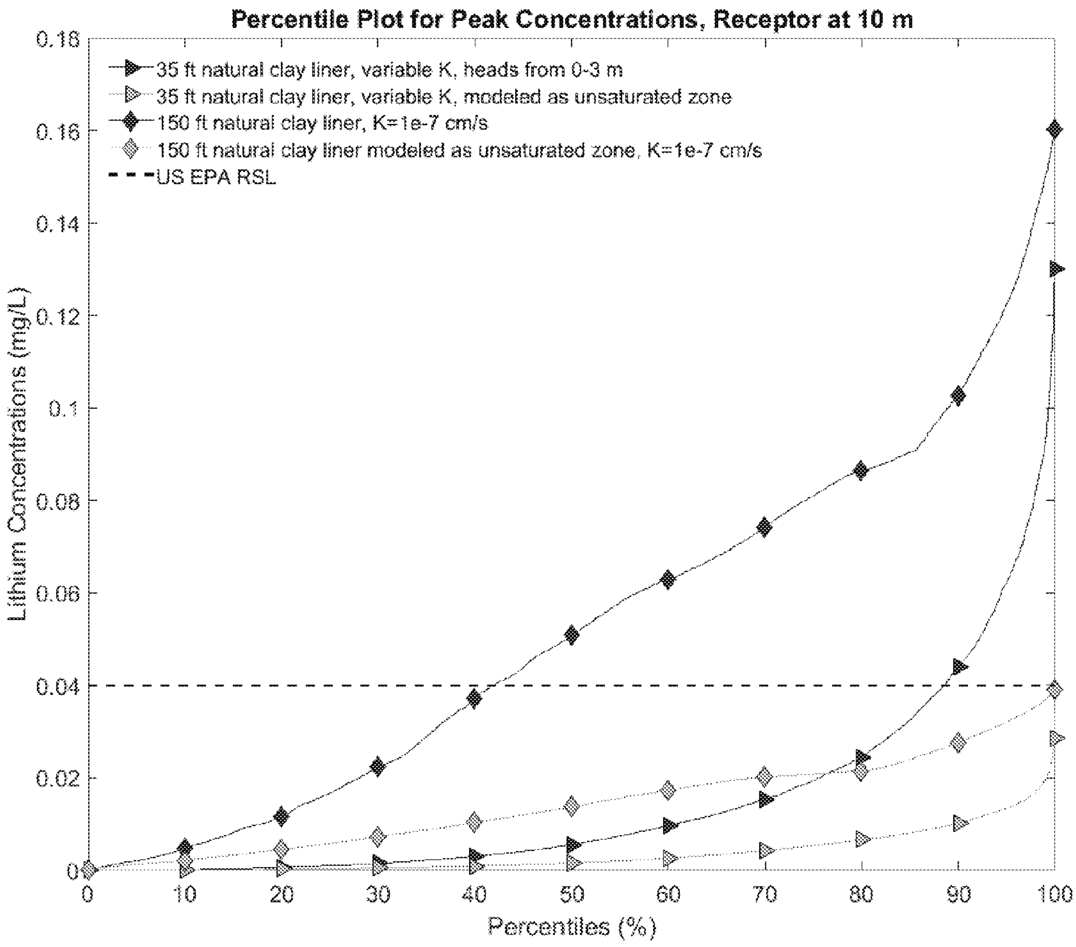

Figure 4.6 Sensitivity Analysis Comparing Different Natural Clay Layer Modeling Approaches – Lithium

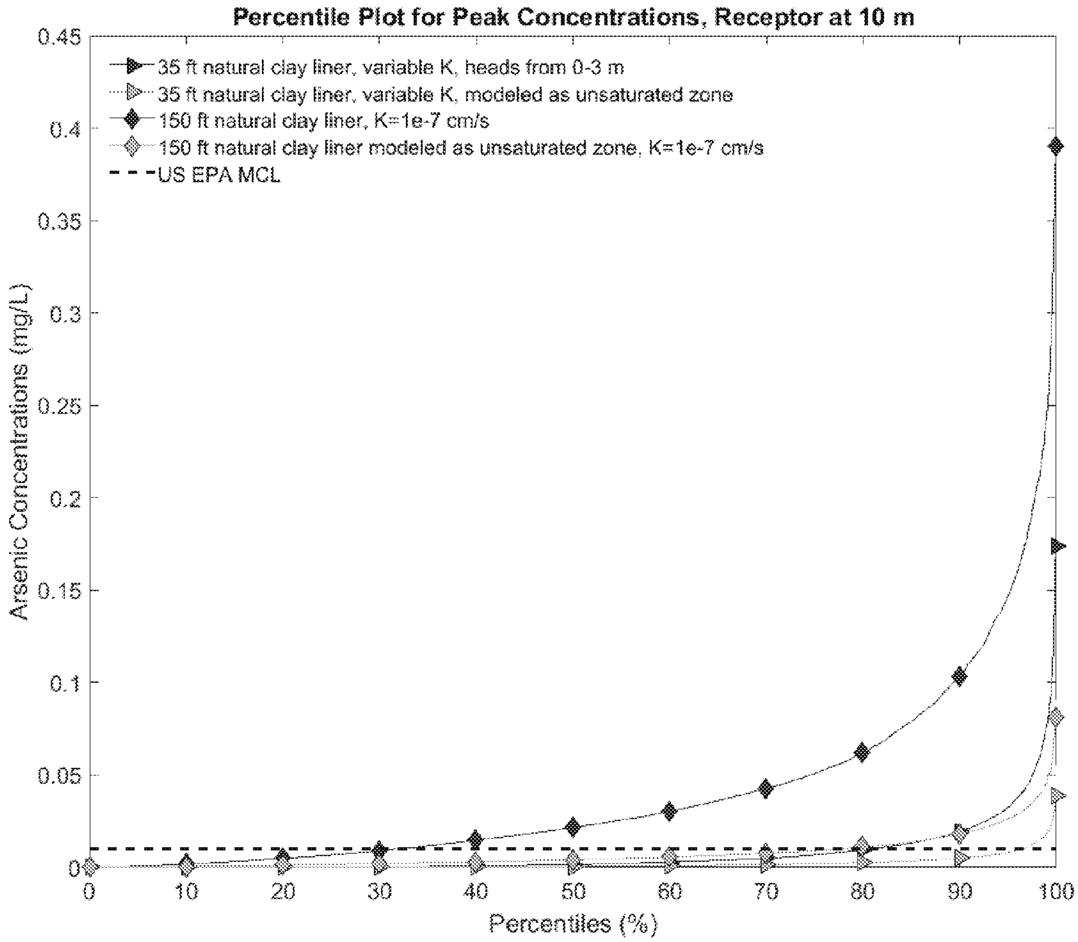

Figure 4.7 Sensitivity Analysis Comparing Different Natural Clay Layer Modeling Approaches – Arsenic

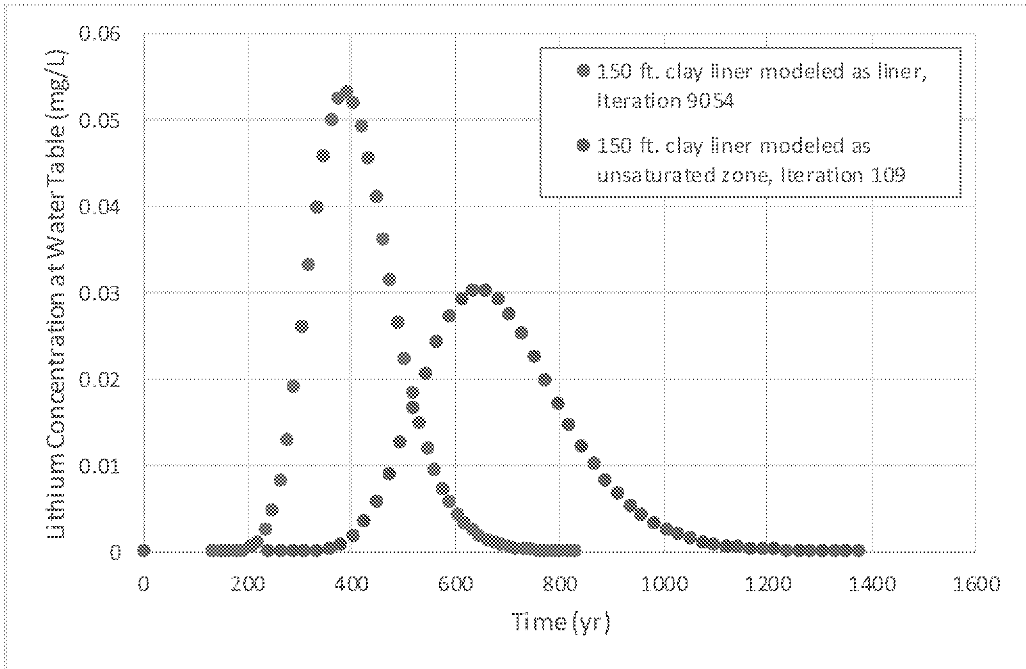

Figure 4.8 Time vs concentration curves showing difference in EPACMTP results when the natural clay is simulated using EPACMTP's liner function (blue) versus simulation as the unsaturated zone (red)

4.2.3 Sensitivity Analysis – Impact of Unconfined Groundwater Conditions at Base of Natural Clay Liner

The natural clay modeling approach in which the natural clay liners were modeled assuming that the clays were part of the liner system (Figure 2.3; Section 4.1), evaluated in Section 4.2.2, also created an additional limitation in that groundwater conditions underlying the natural clay liner were required to be unconfined, with no positive pressure head at the base of the natural clay liner. This limitation is due to a restriction in EPACMTP that precludes groundwater or a groundwater mound from intersecting a model-specified liner. However, changing the modeling approach so that natural clay liners are simulated as part of the unsaturated zone also allows the impact of confined versus unconfined conditions underlying the natural clay liners to be assessed. Confined groundwater conditions underlying a natural clay liner (which is a plausible condition that may occur in reality, because the natural clays frequently act as a confining layer [see Section 5 and Figure 5.1]) will cause a reduction in infiltration across the natural clay liner due to the reduced hydraulic gradient. Figure 4.9 presents a schematic depiction of this scenario.

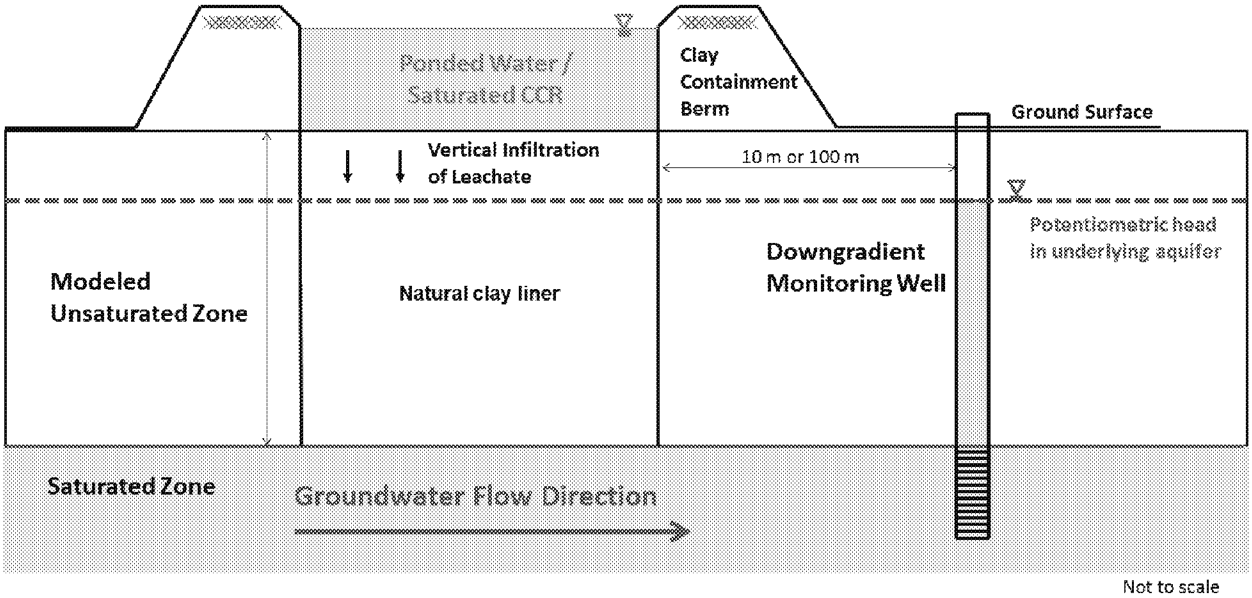

Figure 4.9 Sensitivity Analysis – Scenario with Confined Underlying Groundwater

A sensitivity analysis to assess the conditions depicted in Figure 4.9 was created by modifying the modeled infiltration rates described in Section 4.2.2 for the 35-ft natural clay liner with variable hydraulic conductivity. The presence of a non-zero potentiometric head at the bottom of the liner can be accounted for in the Darcy's Law approach described in Section 3.2.1 by subtracting the potentiometric head at the bottom of the liner (h_a) from the total head term in Equation 2. It was assumed that the potentiometric head in the underlying groundwater aquifer (h_a) was 80% of the thickness of the natural clay liner ($0.8\ t_s$)

$$q = k_s\left(\frac{h - h_a}{t_s} + 1\right) \qquad \text{Eq. 4}$$

The results of the sensitivity analysis are presented in Figures 4.10 and 4.11. Accounting for the pressure head at the base of the natural clay reduces the model-predicted maximum concentrations for both lithium and arsenic relative to scenarios that do not account for the pressure head. Even at the conservative 90th percentile for this scenario, model-predicted concentrations for lithium and arsenic are both below their respective RSL and MCL reference lines.

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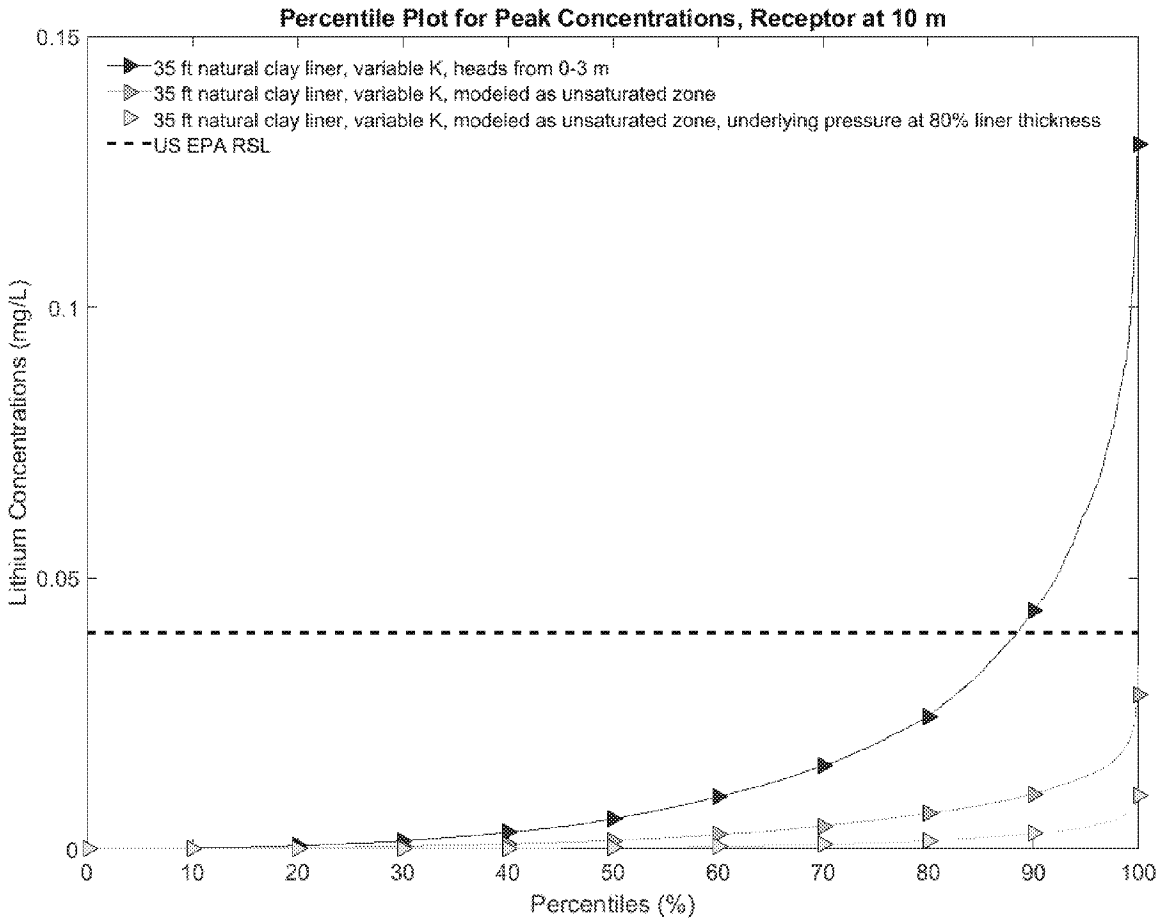

Figure 4.10 Sensitivity Analysis of Results to Confined Underlying Groundwater – Lithium

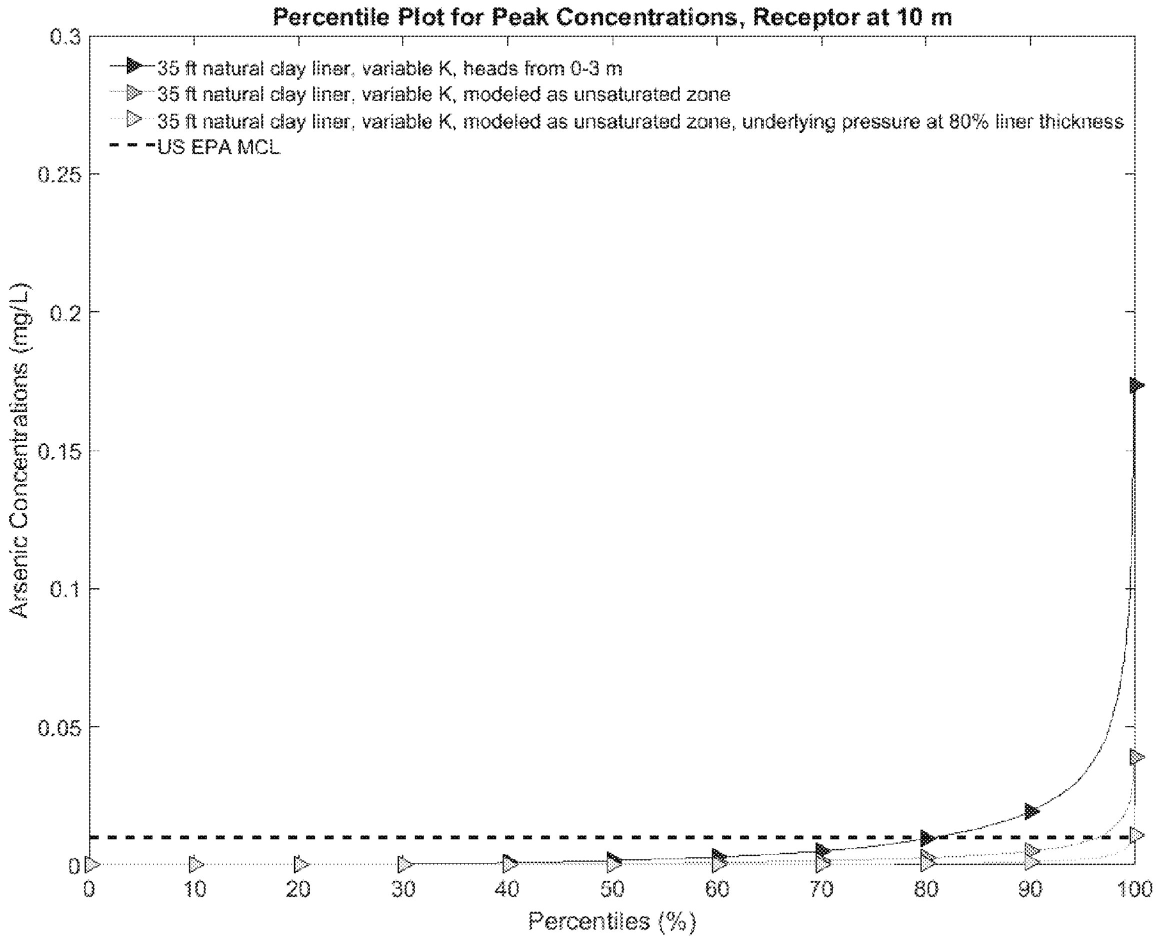

Figure 4.11 Sensitivity Analysis of Results to Confined Underlying Groundwater – Arsenic

5 Case Study

Two CCR SIs (Ponds 1 and 2) located at the JR Whiting Power Plant (JRWPP) in southeastern Michigan were constructed in a natural clay hydrogeologic environment having low hydraulic conductivity. The total area of both Ponds 1 and 2 is about 15 acres. The ponds were operational for 64 years, from 1952 to April 2016. In this section, we evaluate the geologic setting of the JRWPP's CCR ponds using both regional literature and site-specific data analyses to evaluate whether sufficient data have been collected to characterize the natural clays at the JRWPP site and to determine whether the natural clays are operating as an effective liner. The site-specific reports cited in this case study are available for reference as indicated in Appendix C. Moreover, we review the available groundwater quality monitoring data and statistical evaluations to further evaluate whether there is evidence of any preferential flow pathways or leakage through the natural clay liner.

5.1 Geologic Setting

The JRWPP is located in the southeastern portion of Monroe County, Michigan, and is within the Lake Erie-Lake St. Clair basin. The geology, hydrology, and resulting aquifer water quality in this area have been well documented by the United States Geological Survey (USGS, 1996, 1997). In particular, the geology and water quality of Monroe County have been thoroughly described using data from 32 USGS observation wells (USGS, 1996). The deposits present in the southeastern portion of Monroe County are characterized as glacial till underlain by bedrock from the Salina Group (as observed in USGS wells G32 and G33; USGS, 1996). The USGS reports on Monroe County provide a generalized description of the area's geologic setting, while recognizing that the complex depositional histories associated with glacial till can result in significant variations locally. Detailed geologic information specific to the JRWPP site area has been obtained by Consumers Energy to further bolster the site geological characterization.

5.1.1 Glacial Till

The glacial deposits in southeastern Michigan are clay-rich due to their origin from the scouring of the local shales (USGS, 1997, p. 19). These fine-grained, stratified glacial deposits are comprised of clay, silt, and very fine sands, and may include thin, discontinuous layers of sand and gravels (USGS, 1997, Figure 7, p. 50). The glacial deposits have hydraulic conductivities ranging from 3.5×10^{-10} to 3.5×10^{-7} cm/s (USGS, 1997, Figure 7, p. 50). Surficial clay deposits with an approximate depth of 50 ft are found in southeastern Monroe County, near the JRWPP site (USGS, 1996, Figures 6 and 7).

The stratigraphy of the geologic formation adjacent to the JRWPP CCR ponds is described in the boring logs collected during the installation of six monitoring wells along the perimeter of Ponds 1 and 2 (Arcadis of Michigan, 2016, Appendix A). The boring logs describe medium- to high-plasticity clay beginning between 13 and 30 feet below ground surface (ft bgs) and extending to 70 or 80 ft bgs, with some isolated observations of silts, coarse sand, and pebbles. The lithology from 6 ft (where observations begin) to 30 ft bgs is variable, with observations of coal and fly ash, organic materials (roots and peat), and silts. Based on boring log observations (Arcadis of Michigan, 2016, Appendix A), the glacial till layer at the JRWPP site ranges from approximately 40 to 60 ft thick, but the glacial till clay layer underneath Ponds 1 and 2 is estimated to be at least 35 ft thick (limestone bedrock is located at approximately 520 feet above mean sea level [ft AMSL] and the bottoms of Ponds 1 and 2 are located at approximately 555 ft AMSL

[TRC, 2018a]). The reduced glacial till clay thickness under Ponds 1 and 2 is because clay was excavated during pond construction and the ponds were built into the till layer (Figure 5.1). The boring logs include descriptions of thin layers with sand, cobbles, and pebbles in several of the locations, but they are not interpreted as a continuous feature at the site (Arcadis of Michigan, 2016). Furthermore, no fractures in the glacial till are noted in the boring logs.

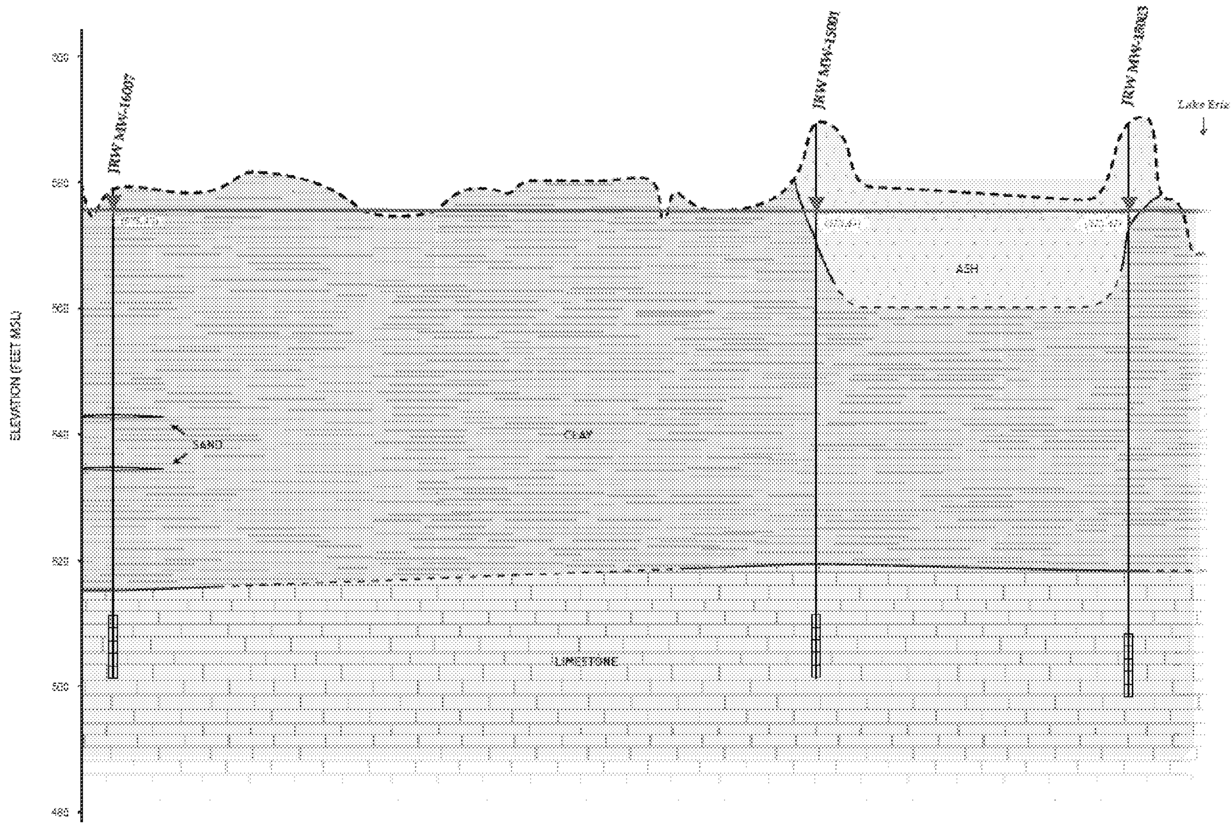

Figure 5.1 Geologic Cross-Section Through Pond 1 – JRWPP Site. Source: TRC (2018a). The blue line indicates the potentiometric surface of the limestone bedrock unit, rather than the water table.

Permeameter tests were conducted on eight samples of clay collected from seven boreholes located elsewhere on the JRWPP site (TRC, 2018a). Each sample represented a vertical interval of approximately 1 ft and was collected at depths between 31.5 and 53 ft bgs, with most samples falling between 33 and 39 ft bgs. The measured hydraulic conductivity values ranged from 5.5×10^{-9} to 2.2×10^{-8} cm/s (TRC, 2018a).

5.1.2 Bedrock

The bedrock surface within Monroe County generally slopes toward Lake Erie and is found at approximately 550 ft AMSL along Lake Erie (USGS, 1996, p. 8). In the southeastern portion of the county, only the Salina Group, the oldest rock unit in the area, is present (USGS, 1996, p. 7). Carbonates dominate the Salina Group, and the bedrock is comprised of limestone and dolomite, with interbedded shales and sulfate-rich evaporite deposits in the area surrounding the JRWPP (anhydrite and gypsum; USGS, 1997, p. 12).

The bedrock aquifer is confined by the surficial clays in southeastern Monroe County near the JRWPP, and groundwater is derived from secondary features (fractures and bedding-plane partings) in the carbonates (USGS, 1996, pp. 16, 20). The main recharge area for this aquifer is within the central portion of the county and the potential for recharge is lowest in the southeastern portion of the county (USGS, 1996, p. 20). Historically, the area near to the JRWPP was a "flowing-well district," indicating that the potentiometric water level was higher than the ground surface (USGS, 1996, p. 2), although the potentiometric surface at the site suggests that this is no longer the case for the area around JRWPP (Figure 5.1).

The bedrock aquifer at the JRWPP site is described in the boring logs collected during the installation of six monitoring wells at the perimeter of the ponds (Arcadis of Michigan, 2016, Appendix A). The bedrock is characterized as a homogeneous, fine-grained limestone with visible ("little large") pores and calcite crystals beginning between 70 and 80 ft bgs and extending the depth of the borehole (typically an additional 15-20 ft bgs) (Arcadis of Michigan, 2016, Appendix A).

Hydraulic conductivity was measured in three of the six monitoring wells using the pneumatic slug test method. The values ranged from 7.1×10^{-3} to 3.5×10^{-3} cm/s (Arcadis of Michigan, 2016, Table 3). Groundwater elevation data collected nine times between December 2016 and October 2017 show that the hydraulic gradient within this aquifer is very low (incalculable) and potentially stagnant; thus, there is no dominant overall flow direction (TRC, 2018a, p. 6).

5.2 Groundwater Monitoring Network

5.2.1 Well Placement

At the JRWPP CCR ponds, the groundwater monitoring system consists of six monitoring wells screened in the confined bedrock aquifer, which are located along the perimeter of the ponds (Figure 5.2). The well screens are 10 ft long and start approximately 5 ft from the top of the bedrock layer; the screen intervals are all found between 78 and 96 ft bgs (Arcadis of Michigan, 2016, Table 1). As described above, the groundwater in the bedrock aquifer is physically separated from the ponds by at least 35 ft of clay-rich glacial till deposits. Water level monitoring in the six wells has demonstrated little to no sustained hydraulic gradient between the wells and no dominant flow direction; thus, a hydraulically upgradient location could not be conclusively identified.

Given the lack of observed discontinuities in the till and bedrock, small (15-acre) area of the site, and the CSM for a natural clay liner that indicates an area-wide release, rather than a pinhole release, the use of six monitoring wells along the perimeter of the ponds is adequate for characterizing groundwater composition and for compliance monitoring. A minimum of three wells are required under the CCR Rule. The additional wells at the JRWPP site address both uncertainties about aquifer heterogeneity and groundwater flow direction. The groundwater monitoring system, with the six wells taken together, provides a thorough representation of the geologic formation on which the ponds are constructed and is positioned to detect changes in groundwater quality within the uppermost aquifer.

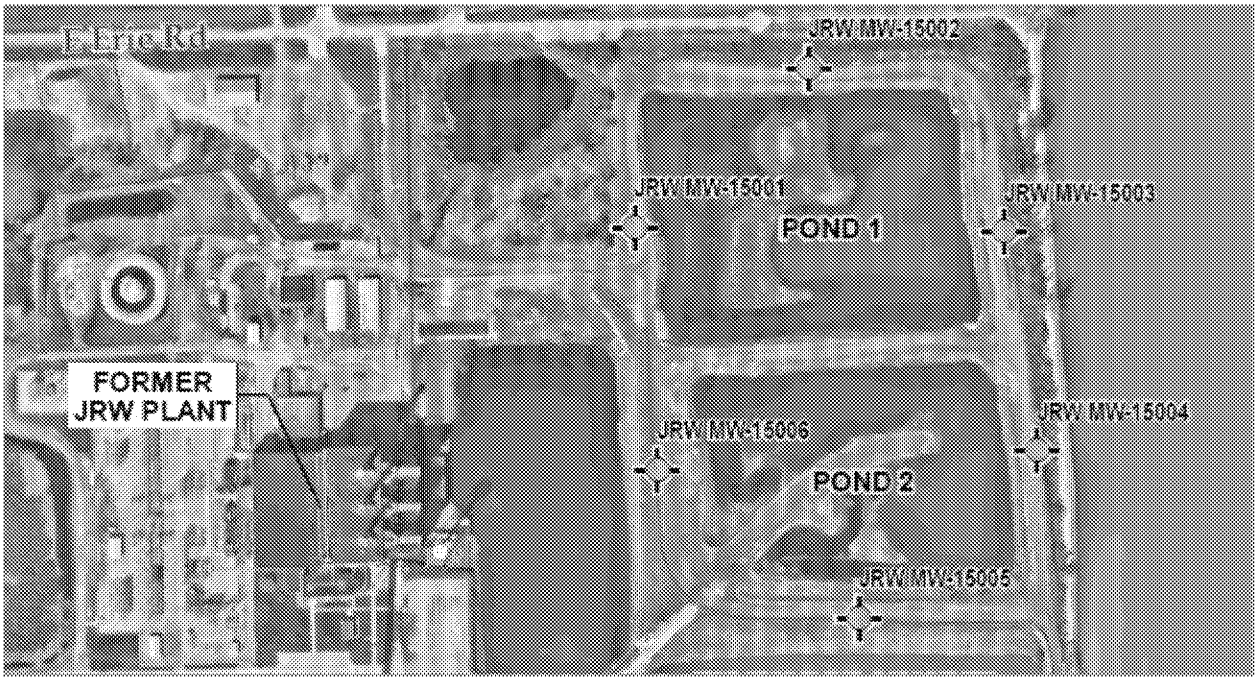

Figure 5.2 Groundwater Monitoring Locations – JRWPP Site. Source: TRC (2018b). Note that since this photo was taken the ponds have been dewatered and final cover is scheduled for 2019.

5.2.2 Regional Water Quality

The bedrock aquifers in Monroe County yield groundwater with highly variable quality due to the bedrock and hydrologic variability of the aquifer (USGS, 1996, p. 32). For example, the total dissolved solids (TDS) concentrations in the carbonate aquifers (Salina Group and others) range from 203 to 2,700 mg/L (USGS, 1996, p. 36, 1997, p. 75). Groundwater in this area with TDS less than 500 mg/L is typically calcium-bicarbonate type, while groundwater with TDS concentrations over 500 mg/L is calcium-bicarbonate-sulfate (mixed anion) to calcium-sulfate (sulfate) type (USGS, 1996, p. 32). In southeastern Michigan, the bedrock aquifer produces sulfate-type water and may have TDS concentrations over 1,000 mg/L (Table 5.1; USGS, 1996, p. 32, Figure 24).

Regional water quality measurements indicate that the shallow bedrock aquifer produces sulfate-type water with high dissolved solids that is typically not considered suitable for drinking. The natural state of this groundwater is that it has cation and anion concentrations that exceed state and federal standards for drinking water, including for chloride, sulfate, and TDS.

5.2.3 Background Water Quality Conditions

Background conditions are typically estimated using upgradient wells and are used to approximate the range of naturally occurring concentrations present at a site. Groundwater monitoring of the six perimeter wells at the JRWPP site was used to establish background conditions, considering data from nine sampling rounds over an 11-month period (from December 2016 to October 2017; TRC, 2018b). These initial results showed that the groundwater present in the bedrock aquifer has elevated concentrations of sulfate, chloride, and TDS, as well as elevated pH (i.e., above US EPA and Michigan drinking water standards [US EPA, 2018c; MDEQ, 2018]), similar to the observations described in the USGS reports for Monroe County

(USGS, 1996, 1997), and low (<250 µg/L) concentrations of boron (Figure 5.3). Boron is a key indicator constituent for coal ash leachate (EPRI, 2012).

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Table 5.1 Groundwater Quality in Southeastern Michigan

Parameter	Units	Criteria				Groundwater Data			
		US EPA MCL	MI Residential	MI Non-Residential	MI GSI	Southeast Michigan	Monroe County	USGS Well Nearest to JRWPP	Groundwater Monitoring at JRWPP
Appendix III Constituents									
Boron	µg/L	NC	500	500	7,200	NR	NR	NR	166-247
Calcium	mg/L	NC	NC	NC	500	57-400	29-460	220	87.1-165
Chloride	mg/L	250[a]	250	250	**50**	9.4-10	**1.1-600**	36	**35.7-52.5**
Fluoride	µg/L	4,000	NC	NC	NC	NR	100-2,700	2,700	1,030-1,700
pH, Field	SU	**6.5-8.5**	**6.5-8.5**	**6.5-8.5**	6.5-9.0	NR	7.0-8.0	7.4	**7.4-8.7**
Sulfate	mg/L	**250[a]**	**250**	**250**	500	**150-1,200**	**<0.1-1,400**	**800**	**282-469**
Total Dissolved Solids	mg/L	**500[a]**	**500**	**500**	**500**	**207-2,430**	**203-2,700**	**1,270**	**610-984**
Appendix IV Constituents									
Antimony	µg/L	6	6	6	130	NR	NR	NR	All ND (<1.0)
Arsenic	µg/L	10	10	10	10	NR	<1-1	<1	<1-3.2
Barium	µg/L	2,000	2,000	2,000	670	NR	6-320	24	9.9-40
Beryllium	µg/L	4	4	4	6.7	NR	<0.5-2	0.5	All ND (<1.0)
Cadmium	µg/L	5	5	5	3	NR	All ND (<5.0)	<1.0	<0.2-0.2
Chromium	µg/L	100	100	100	100	NR	<1-5	<1.0	<1-3.7
Cobalt	µg/L	NC	40	100	100	NR	All ND (<20)	<3.0	<15-15
Fluoride	µg/L	4,000	NC	NC	NC	NR	100-2,700	2,700	1,030-1,700
Lead	µg/L	NC	4	4	29	NR	<1-7	<1.0	<1-1
Lithium	µg/L	NC	170	350	440	NR	<4-200	55	39-71
Mercury	µg/L	2	2	2	0.2	NR	NR	NR	<0.2-0.2
Molybdenum	µg/L	NC	73	210	3,200	NR	All ND (<30)	<10	<5-6
Radium-226	pCi/L	5	NC	NC	NC	NR	NR	NR	0.271-2.46
Radium-226/228	pCi/L	5	NC	NC	NC	NR	NR	NR	0.396-4.37
Radium-228	pCi/L	5	NC	NC	NC	NR	NR	NR	0.338-3.48
Selenium	µg/L	50	50	50	5	NR	NR	NR	<1-1
Thallium	µg/L	2	2	2	3.7	NR	NR	NR	<2 - <20

Notes:

GSI = Groundwater-Surface Water Interface; JRWPP = JR Whiting Power Plant; MCL = Maximum Contaminant Level; MI = Michigan; NC = No Criteria; ND = Not Detected; NR = Not Reported; US EPA = United States Environmental Protection Agency; USGS = United States Geological Survey.

BOLD value indicates at least one exceedance of the listed criteria.

Most metals are reported as dissolved in USGS reports (USGS, 1996, 1997). JWRPP data are analyzed as total metals unless otherwise specified.

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Sources:
US EPA MCL = US EPA (2018c).
MI Residential, Non-Residential, and GSI = Michigan Part 201 Generic Drinking Water Cleanup Criteria (MDEQ, 2018).
Southeast Michigan = Values from the Carbonate Aquifer in Southeast Michigan (USGS, 1997, Table 17).
Monroe County = Values from the Silurian-Devonian Aquifer (USGS, 1996, Table 4).
USGS Well Nearest to JRWPP = Values from USGS Well 33 (USGS, 1996, Table 2).
Groundwater Monitoring at JRWPP = Includes results from 10 sampling events of 6 wells (JWR-MW-15001 to JWR-MW-15006) between December 2016 and November 2017 (TRC, 2018b). Appendix IV constituents were not reported for the November 2017 event.
(a) Secondary MCL.

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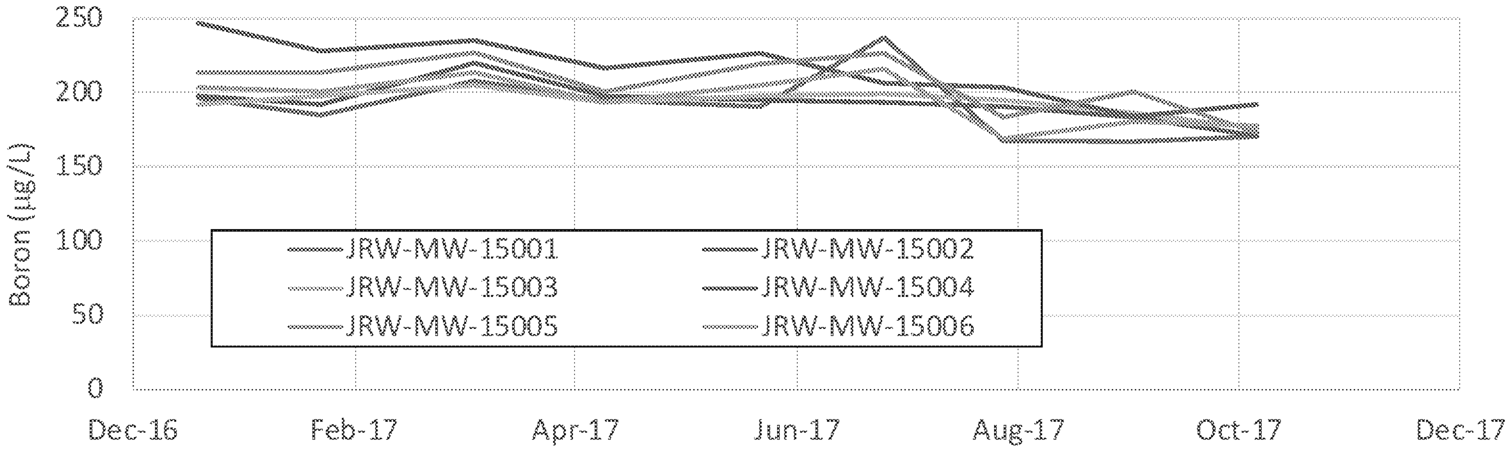

Figure 5.3 Boron Concentrations Over Time at On-Site Monitoring Wells – JRWPP Site. Source: TRC (2018a).

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Based on the lack of a dominant flow direction at the JRWPP site and the naturally elevated concentrations of many constituents, an intrawell, prediction interval approach was selected by TRC as the most appropriate statistical procedure for evaluating potential compliance at these wells (TRC, 2017). This approach compares concentrations measured at a well against a series of background measurements from the same well. Boron concentrations are low, and do not exhibit an increasing trend (167-247 µg/L; Figure 5.3), indicating that groundwater has not been impacted by a release from the JRWPP CCR ponds and that intrawell statistics are acceptable for this site. Prediction limits for each well are established by estimating the upper limit of future values based on the initial observations of the range of naturally occurring concentrations at the site. TRC established prediction limits for the seven constituents listed in Appendix III of the CCR Rule for each well to analyze results from the nine initial sampling rounds. The prediction limits for each constituent and well are shown in Table 5.2 (TRC, 2018b) and may be revised periodically (every four to eight sampling rounds; TRC, 2017).

Table 5.2 Prediction Limits of Appendix III Constituents – JRWPP Site

Constituent	JRW-MW-15001	JRW-MW-15002	JRW-MW-15003	JRW-MW-15004	JRW-MW-15005	JRW-MW-15006
Boron (µg/L)	251	229	219	271	256	240
Calcium (mg/L)	182	185	162	143	127	144
Chloride (mg/L)	54.4	54.5	55.5	54.7	44	52.1
Fluoride (mg/L)	1,560	1,870	1,810	1,860	1,730	1,710
Field pH (SU)	7.4-8.1[a]	7.3-7.8	7.4-8.2	7.4-7.9	7.7-8.4	7.1-9
Sulfate (mg/L)	469	495	454	389	347	404
Total Dissolved Solids (mg/L)	974[a]	1,020	969	900	844[a]	922[a]

Notes:

JRWPP = JR Whiting Power Plant.

(a) Indicates that a non-normal distribution was used. For the total dissolved solids non-normal distributions, the maximum of the nine background values was used. For the field pH, the minimum and maximum of the nine background values were used (TRC, 2018b).

5.2.4 Detection Monitoring Events and Statistically Significant Increases

As mandated by the CCR Rule, the results from ongoing detection monitoring events are evaluated against the prediction limits established using the background sampling. If a new result does not fall within the established prediction limits, the result is tentatively identified as a statistically significant increase (SSI) and requires further evaluation within 90 days. The results for the first round of detection monitoring are provided in Table 5.3 (December 2017; TRC, 2018b). Potential SSIs were identified for field pH measurements in three wells (with the pH being lower than the established prediction limits). These wells were resampled within 30 days for pH alone and no exceedances of the prediction limits were found upon resampling; therefore, no SSIs were recorded for this detection monitoring event (TRC, 2018b).

Table 5.3 Detection Monitoring Results, November 2017 – JRWPP Site

Constituent	JRW-MW-15001	JRW-MW-15002	JRW-MW-15003	JRW-MW-15004	JRW-MW-15005	JRW-MW-15006
Boron (μg/L)	179	187	176	207	173	166
Calcium (mg/L)	128	137	114	103	90.5	102
Chloride (mg/L)	51.9	50.6	49	52.5	40.5	49.2
Fluoride (mg/L)	1,400	1,500	1,500	1,400	1,300	1,400
Field pH (SU)[a]	6.8 (7.5)	7 (7.6)	7.4	7.2 (7.7)	7.9	7.7
Sulfate (mg/L)	439	464	390	356	325	373
Total Dissolved Solids (mg/L)	934	832	758	686	644	700

Notes:
JRWPP = JR Whiting Power Plant.
Samples were collected on November 13, 2017.
Data reported in TRC (2018b).
(a) The field pH measurements in 3 wells were below the established prediction limits and required verification resampling, which occurred on January 18, 2018. Results from the re-sampling are given in parentheses, and these results were all within the prediction limits.

5.3 Summary

The USGS reports for Monroe County describe a carbonate bedrock aquifer overlain by a surficial glacial clay layer that is regionally extensive, with a typical thickness of 50 ft in southeastern Michigan (USGS, 1996, 1997). At the JRWPP site, the clay thickness has been observed to be at least 35 ft from the bottoms of Ponds 1 and 2 to the bedrock aquifer. Hydraulic conductivity measurements of on-site clays range from approximately 10^{-9} to 10^{-8} cm/s. The geology between boreholes is consistent, and no fractures or continuous sand strings were identified that could serve as continuous flow paths in the clay.

Groundwater quality is being monitored at the JRWPP site using six perimeter wells located near the CCR ponds (TRC, 2018b). The placement of the wells is near the top of the bedrock aquifer and along the perimeter of the ponds, in order to detect changes in groundwater quality in this area of near-stagnant groundwater flow. Nine sampling events were used to characterize the range of concentrations at the site, and the variability of the concentrations was statistically evaluated to establish prediction limits for each constituent in each well (TRC, 2018b). During the background monitoring period, groundwater contained elevated concentrations of sulfate, major cations, and TDS, similar to the wells described in the USGS reports on Monroe County (USGS, 1996, 1997). During detection monitoring that has been completed to date, no SSIs have been reported (TRC, 2018b). The key finding for this evaluation is that boron concentrations in the uppermost aquifer are low (167-247 μg/L) after 64 years of operation. The lack of elevated boron concentrations indicates that SI operations have not had an impact on groundwater quality at this site.

Overall, the geology at the JRWPP site, and specifically underlying CCR Ponds 1 and 2, is well characterized, consisting of a continuous layer of low-permeability clays. These clays are serving as a natural clay liner, separating the CCR ponds from the underlying bedrock aquifer. The groundwater monitoring system has six monitoring wells and is positioned to detect changes in groundwater quality. To date, no groundwater concentration impacts associated with CCR Pond 1 or 2 have been detected at any of the JRWPP site's groundwater monitoring wells.

6 Summary and Conclusions

The objective of the research described in this white paper was to perform modeling and document a case example as part of an evaluation of whether or not an alternative SI liner can perform similarly to a composite liner as defined under the CCR Rule, and by extension be protective of HHE. This research is supplemented with a conceptual evaluation of SI liner differences, presented in a separate white paper (Benson 2019).

This study presents comparative probabilistic modeling of the performance of different liners in use at SI sites using US EPA's EPACMTP model, along with estimates for liner infiltration rates widely known and used by practitioners. Two risk-driving analytes, lithium and arsenic, were simulated. The modeling approach was consistent with US EPA's CCR Risk Assessment (US EPA, 2014), when feasible, and reasonable parameter selections were made for other inputs in order to facilitate a relative evaluation of liner performance.

The base case for the modeling was a composite liner consisting of a 60-mil HDPE layer over a 2-ft compacted soil layer with a hydraulic conductivity of 10^{-7} cm/s. An unlined scenario was also included to represent an uncontrolled scenario. Alternative liners that were simulated included an engineered clay liner, several variations of thick natural clay liners, and composite liners that differ from the criteria specified in the CCR Rule. The results were evaluated by comparing maximum lithium and arsenic concentrations predicted by the model for various liner scenarios. Health-based values (specifically, the RSL for lithium and the MCL for arsenic) are also presented for reference, although it is important to note that there are numerous conservative assumptions used in this modeling such that any comparison to the health-based numbers is also conservative.

The results of the comparative modeling show that the alternative composite liners that were simulated performed similarly to the base case composite liner. In addition, thick natural clay liners with a hydraulic conductivity of about 10^{-8} cm/s performed more similarly to the base case composite liner scenario than to the unlined scenario. Conversely, engineered liners and thick natural clay liners with a hydraulic conductivity of 10^{-7} cm/s yielded results that were closer to the unlined scenario than to the base case composite scenario, although these liner alternatives were less likely than the unlined scenario to yield a constituent concentration higher than the reference health-based criteria. Sensitivity analyses showed that more refined modeling scenarios for natural clay liners that account for constituent migration and sorption within the liner in addition to potential confined groundwater conditions in the underlying aquifer reduced the maximum model-predicted downgradient concentrations; for these revised scenarios, performance of the natural clay liners was closer to that predicted for the baseline composite scenario.

Even with the conservative assumptions used in this modeling, thousands of combinations of environmental parameters identified in the Monte Carlo analysis for each scenario yielded model results in which maximum constituent concentrations at the point of compliance did not exceed the human health benchmark within the 10,000-year timeframe of the modeling. These results, by extension, suggest that there are many plausible scenarios in which alternative liners can be protective of HHE.

Highlighting the previous point, a case study was presented that shows an effective natural clay liner at an SI site in southeastern Michigan. The geology has been well characterized both by regional USGS studies and by site-specific investigations showing that a competent 35-ft-thick natural clay layer with a hydraulic conductivity between 5.5×10^{-9} and 2.2×10^{-8} cm/s is present underlying the SI. A groundwater monitoring

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network has been established that shows low, stable concentrations of boron, the key indicator constituent for CCR leachate, and no SSIs for other monitored constituents.

Taken together, the results of the modeling and the case study provide evidence that certain alternative liners, such as non-federally compliant composite liners and thick natural clay liners with low hydraulic conductivity, can achieve performance approaching that of the base case composite liner, which the US EPA (2014) has determined is protective of HHE.

This finding is not intended to support use of alternative liners for new units. As noted by Benson (2019) in the companion white paper, the base case composite liner is protective regardless of hydrogeologic environment. Rather, these results suggest that certain existing units with non-federally compliant composite liners, or with thick natural clay liners that have low hydraulic conductivity, can be similarly protective of HHE as the base case composite at sites with favorable hydrogeologic conditions, which can be established using a performance standard approach to review groundwater monitoring data at these facilities to assure that there are low boron concentrations in groundwater and no SSIs caused by SI operation.

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Appendix A

EPACMTP Model Input Files

Due to their large file size, the model files are not included with this document. Please contact Bruce Hensel (bhensel@epri.com) if interested in obtaining the model input files.

Appendix B

Tabulated Modeling Results

Due to their large file size, the model files are not included with this document. Please contact Bruce Hensel (bhensel@epri.com) if interested in obtaining the model output files.

Appendix C

Case Study Supporting Documentation

Due to large file size, the case study supporting documentation is not included with this document. Please contact Bruce Hensel (bhensel@epri.com) if interested in obtaining the documentation.

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